

Chapter 3 Affected Environment

Section 1502.15 of the CEQ NEPA regulations (40 CFR parts 1500-1508) direct that “an environmental impact statement shall succinctly describe the environment of the area(s) to be affected or created by the alternatives under consideration. The descriptions shall be no longer than is necessary to understand the effects of the alternatives. Data and analyses in a statement shall be commensurate with the importance of the impact, with less important material summarized, consolidated, or simply referenced. Agencies shall avoid useless bulk in statements and shall concentrate effort and attention on important issues. Verbose descriptions of the affected environment are themselves no measure of the adequacy of an environmental impact statement.”

This affected environment section contains succinct descriptions of the resources and issues identified as important to predicting the direct, indirect, and cumulative impacts that will accrue from the proposed action. The topics in this section are arranged in the same order they are treated in the environmental consequences section that follows: marine mammals, target fish species, non-specified fish species, forage fish, prohibited species, ESA listed Pacific salmon, seabirds, marine benthic habitat, ecosystem, State managed fisheries, management complexity and enforcement issues, and socio-economic issues.

References to original literature are made throughout the section to identify scientific sources and guide readers to further information. All references called out throughout chapters 1, 2, 3, and 4 of this document are listed in chapter 7 of this document. Each reference contains information necessary to find the respective paper, report, journal article, or book, following standard library citation format. Any reader desiring to access one of the references given should be able to read or borrow a copy from a public library. Resource libraries located in Alaska and the Northwest such as the NMFS Alaska Fisheries Science Center Library in Seattle, Washington, or the Auke Bay Fisheries Laboratory near Auke Bay, Alaska, the State Library in Juneau, Alaska, and the libraries at the University of Alaska and the University of Washington, will be more likely to have these particular references on their shelves. Other libraries are able to get these references through interlibrary loan systems.

3.1 Marine Mammals

Because of combinations of bathymetric and oceanographic features, the Bering Sea and Aleutian Island (BSAI) and Gulf of Alaska (GOA) are among the world’s most productive ecosystems, and supports a rich assemblage of marine mammals (Loughlin, *et al.*, 1999). Marine mammals occur in diverse habitats, including deep oceanic waters, the continental slope, and the continental shelf (Lowry *et al.*, 1982). In the areas fished by commercial groundfish fleets, twenty-six species of marine mammals are present from the orders Pinnipedia (seals, sea lion, and walrus), Carnivora (sea otter and polar bear), and Cetacea (whales, dolphins, and porpoises) (Lowry and Frost 1985). Most species are resident throughout the year, while others seasonally migrate into or out of the management areas. Following are descriptions of their distribution, population status, abundance, habitat, diet, and known sources of anthropogenic mortality.

3.1.1 Steller Sea Lion

The Steller sea lion (*Eumetopias jubatus*), also found in the literature as Steller’s sea lion and northern sea lion, probably evolved in the North Pacific at least 3-4 million years ago (Repenning, 1976). Steller sea lions are members of order the Pinnipedia, family Otariidae (composed of fur seals and sea lions), subfamily Otariinae, a group of five sea lion species (California (*Zalophus californianus*), Steller, southern (*Otaria*

bryonia), Australian (*Neophoca cinerea*), and New Zealand (*Phocartos hookeri*)) mostly distributed in temperate and subpolar waters.

3.1.1.2 Distribution

The Steller sea lion ranges along the North Pacific Ocean rim from northern Japan to California (Loughlin *et al.*, 1984), with centers of abundance and distribution in the GOA and Aleutian Islands, respectively (Figure 3.1-1). The northernmost breeding colony in the Bering Sea is on Walrus Island near the Pribilof Islands, and in the GOA on Seal Rocks in Prince William Sound, the northern most of all sea lion rookeries (Kenyon and Rice, 1961).

In general sea lions seem to have a high degree of site fidelity, returning to breed at or near their natal rookeries (Loughlin, 1997; Raum-Suryan *et al.*, submitted). Tagged and branded individuals have been seen at distances up to 1784 km from their natal rookeries, but once they approach adulthood they generally remain within 500 km of their natal rookery (Raum-Suryan *et al.*, submitted).

3.1.1.3 Population Status and Trends

In November 1990, the NMFS listed Steller sea lions as “threatened” range-wide under the U.S. Endangered Species Act (55 Federal Register 49204, November 26, 1990) in response to a population decrease of 50% - 60% during the previous 10 - 15-year period. Several years later, two population stocks were identified, based largely on differences in genetic identity, but also on regional differences in morphology and population trends (Bickham *et al.*, 1996; Loughlin, 1997). The western stock, which occurs from 144° W long. (approximately at Cape Suckling, just east of Prince William Sound, Alaska) westward to Russia and Japan, was listed as “endangered” in June 1997 (62 Federal Register 24345, May 5, 1997). The eastern stock, which occurs from Southeast Alaska southward to California, remains classified as threatened.

3.1.1.3.1 Western Stock

Population assessment for Steller sea lions has been achieved primarily by aerial surveys and on-land pup counts. Historically, this included surveys of limited geographical scope in various portions of the species’ range, in many cases conducted using different techniques, and occasionally during different times of year. Consequently, reconstructing population trends for Steller sea lions from the 1970s and earlier, and over a large geographical area, such as the Western Stock in Alaska, includes a patchwork of regional surveys conducted over many years.

Aerial surveys conducted from 1953 through 1960 resulted in combined counts of 170,000 to 180,000 Steller sea lions in what we now define as the Western Stock in Alaska (Mathisen, 1959; Kenyon and Rice, 1961). Surveys during 1974-1980 suggested an equivocal increase to about 185,000, based on maximal counts at sites over the same area, as summarized by Loughlin *et al.* (1984). It was concurrent with the advent of more systematic aerial surveys that population declines were first observed. Braham *et al.* (1980) documented declines of at least 50% from 1957 to 1977 in the eastern Aleutian Islands, the heart of what now is the Western Stock. Merrick *et al.* (1987) estimated a population decline of about 50% from the late 1950s to 1985 over a much larger geographical area, the central Gulf of Alaska through the central Aleutian Islands, although this still included a patchwork of regional counts and surveys (Figures 3.1-1, 3.1-2). The population in the Gulf of Alaska and Aleutian Islands declined by about 50% again from 1985 to 1989, or an overall decline of about 70% from 1960 to 1989 (Loughlin *et al.*, 1992).

Much of the population trend analyses during recent years has focused on “trend sites” as espoused by the Steller Sea Lion Recovery Team (NMFS 1992b, NMFS 1995a). Trend sites are those rookeries and haul-out sites surveyed consistently from the mid 1980s to the present, thus allowing analysis of population trends on a decadal scale. Trend sites include about 75% of animals observed in recent surveys (Strick *et al.*, 1997; Sease *et al.*, 1999; Sease and Loughlin, 1999; Sease *et al.*, 2001). At 82 rookery and haul-out trend sites in the western stock, the June 2000 count represented declines of 10.3% from 1998, 17.5% from 1996, and 40.0% from 1990 (Figure 3.1-1) (Sease *et al.*, 2001). The average annual rate of decline has been consistently around 5% during the 1990s (Strick *et al.*, 1997; Sease *et al.*, 1999; Sease and Loughlin, 1999; Sease *et al.*, 2001).

The population decline for the Western Stock in Alaska has been apparent in all regions, although not at the same rate. The decline was first observed in the eastern Aleutian Islands (Braham *et al.*, 1980). During subsequent years the decline spread into adjacent regions in the Aleutian Islands and Gulf of Alaska (Merrick *et al.*, 1987). In the eastern Aleutian Islands, the rate of decline lessened and by 1989 or 1990 the population there appeared to stabilize, but at very low levels (Figure 3.1-2). Since 1975 there has been a steady rate of decline of 6% a year or greater, with an additional drop of about 8.7% per year during the late 1980s when the population from the Kenai Peninsula to Kiska Island in the central Aleutian Islands declined at about 15.6% per year (York *et al.*, 1996). Other regions have demonstrated short periods of stability within a general declining trend. With the exception of the differentiation between the eastern and western stocks, however, these regional boundaries are not based on ecological or other biological parameters, and differences in regional trends should be interpreted with caution.

In most years, pups within the Western Stock in Alaska have been counted only at selected rookeries, and on an alternating schedule to minimize potential cumulative effects of disturbance. Range-wide survey efforts included pup counts at virtually all Western Stock rookeries in Alaska in 1998, and all except the Near Islands in the western Aleutian Islands in 1994 (Strick *et al.*, 1997; Sease and Loughlin, 1999). Pup counts in the western stock in Alaska (excluding the western Aleutian Islands) declined by 19.0% from 1994 to 1998. In the western Aleutian Islands, pup numbers declined 18% from 1997 to 1998, the only years for which comprehensive comparison is possible (Figure 3.1-3). Recent pup counts in 2000 and 2001 from Seguam Island to Prince William Sound were similar in magnitude to those conducted in 1998 (NMFS NMML data, unpublished).

3.1.1.3.2 Eastern Stock

Loughlin *et al.* (1992) described Southeast Alaska as the only region of Alaska in which the Steller sea lion population appeared to be stable in 1989, even though numbers of non-pup sea lions (adults and juveniles combined) in Southeast Alaska increased by about 16% from 1985 to 1989, or by an average of 3.5% to 4.0% per year. Calkins *et al.* (1999) estimated that the Steller sea lion population in Southeast Alaska increased by an average of 5.9% per year from 1979 to 1997, based on counts of pups at the three rookeries in the region. From 1989 to 1997, however, pup numbers increased by only 1.7% and counts of non-pups at 12 index sites were stable (average change of +0.5% per year). The Steller Sea Lion Recovery Team employed a different set of index, or “trend,” sites for monitoring population status (NMFS, 1992b; NMFS, 1995a). Counts of non-pup sea lions at the three rookeries and ten haul-outs sites showed an overall increase of 29.3% from 1990 to 2000, or an average annual increase of 1.9% (Sease *et al.*, 2001). Despite differences in individual index sites or model type (e.g., based on counts of pups versus non-pups), the conclusion is that numbers of Steller sea lions in Southeast Alaska are stable or increasing slightly (Figure 3.1-4).

Steller sea lions in Southeast Alaska are not an isolated stock, as demonstrated by genetic data and by the movement of branded and tagged animals from Southeast Alaska to British Columbia and Washington (Raum-Suryan *et al.*, submitted). The number of non-pup sea lions in British Columbia is similar to the number in southeast Alaska, and increasing by about 2.5% per year during the last decade (Figure 3.1-5). Numbers of pups in British Columbia have increased by about 1.5% per year during the same time (personal communication from P. Olesiuk, Pacific Biological Laboratory, Nanaimo, British Columbia, V9R 5K6). Counts of Steller sea lions in Oregon and northern California have been stable during recent decades at about a third as many animals as in either British Columbia or Southeast Alaska. Numbers in central and southern California have been small, but decreasing at about 4.5% - 5.0% per year since 1982 or as much as 10% per year since 1990 (NMFS, 1995; Calkins *et al.*, 1999; Ferrero *et al.*, 2000; Angliss *et al.*, 2001). Despite the observed declines in southern and central California, the Eastern Stock as a whole is stable or increasing slowly.

3.1.1.4 Reproduction and Growth

Steller sea lions have a polygynous reproductive system in which a single male may mate with multiple females. Males establish territories in May in anticipation of female arrival (Pitcher and Calkins, 1981). Mating occurs on land (or in the surf or intertidal zones), thus males are able to defend territories and thereby exert at least partial control over access to adult females and mating privileges. The pupping and mating season is relatively short and synchronous, probably due to the strong seasonality of the environment and the need to balance aggregation for reproductive purposes with dispersion to take advantage of distant food resources (Bartholomew, 1970). In late May and early June, adult females arrive at the rookeries, where pregnant females give birth to a single pup (twinning is rare). Viable births begin in late May and continue through early July. The sex ratio of pups at birth is approximately 1:1, though biased toward slightly greater production of males (e.g., Pike and Maxwell, 1958; Lowry *et al.*, 1982; NMFS, 1992b). Pupping is highly synchronous throughout the sea lion range between the Aleutian Islands and California, with a median pupping date of 12-13 June (Bigg 1985, Merrick 1987). There is evidence that pupping dates have become later at Sugarloaf and Fish Island rookeries (Pitcher *et al.*, 1996). Because timing of pupping depends upon maternal condition, this change may reflect changing environmental conditions, or changing maternal age structure.

Merrick *et al.* (1995) compared pup sizes at different sites where Steller sea lion populations were either decreasing or increasing, to determine if pup size or growth may be compromised in decreasing populations. Their results were not consistent with that hypothesis; rather, they found that pups about two to four weeks of age weighed more at sites in the Aleutian Islands and GOA than in southeast Alaska or Oregon. Fadely and Loughlin (2001) also found that these pups did not weigh greater than expected based on body length, such that they were larger overall. These size differences may arise through different growth rates, as no significant differences have been found among neonatal mass among rookeries (Brandon and Davis, 1999; Adams, 2000). Brandon and Davis (1999) and Adams (2000) found that pups at rookeries in areas of decline grew faster than pups from Southeast Alaska. As there were no differences in milk or energy intake among pups at these rookeries, differences in growth rates may be attributable to differences in pup activity (Adams, 2000), time spent fasting between suckling bouts, or other physiological costs. These observed differences indicate that at least this phase of reproduction may not be affected; that is, if females are able to complete their pregnancy and give birth, then the size of those pups does not appear to be compromised. Possible alternative explanations for the observed size differences are that pups were measured at different ages (i.e., pups in the GOA and Aleutian Islands may have been born earlier and therefore were older when weighed), or that over time, harsher environmental conditions in the Aleutian Islands of the GOA have selected for

larger pup size. Pup condition, measured as the ratio of observed body mass to that expected based on length, seems to be a reasonable index of condition related to survival (Trites and Jonker, 2000). For the pups aged between 2 and 4 weeks, there was no general relationship between pup condition and pup numbers or magnitude of decline at rookeries, though the poorest average pup conditions during the late 1990s were associated with areas of greatest decline (Fadely and Loughlin, 2001). There also was evidence that pup condition was poorest during weak depressions of the Aleutian Low, and better when the Aleutian Low was stronger.

Mothers nurse pups during the day, staying with a pup for the first week, then go to sea on foraging trips. Maternal attendance patterns seem to vary over the range, with the average length of foraging trips during lactation being about 24 hours to two days at the southernmost rookery at Año Nuevo Island, California (Higgins *et al.*, 1988; Hood and Ono, 1997), about 25 hours at Lowrie Island, 19 hours at Fish Island, 11 hours for Chirikof Island, and 7 hours in the Aleutian Islands (Brandon and Davis, 1999). Pups generally are weaned before the next breeding season, but it is not unusual for a female to nurse her offspring for a year or more. The ramifications of nursing a pup beyond a year on pupping frequency, or survival of a second pup, are unknown.

The length of the nursing period may be an important indicator of the female's condition and ability to support her pup, and the pup's condition at weaning (and hence, the likelihood that the pup will survive the post weaning period). Steller sea lion weaning takes place away from the rookeries, over a period of time, and thus has not been directly observed in Alaska. Thorsteinson and Lensink (1962) suggested that nursing of yearlings was common at Marmot Island in 1959. Pitcher and Calkins (1981) suggested that it is more common for pups to be weaned before the end of their first year, but they also observed nursing juveniles (aged 1 - 3). Porter (1997) distinguished metabolic weaning (i.e., the end of nutritional dependence of the pup or juvenile on the mother) from behavioral weaning (i.e., the point at which the pup or juvenile no longer maintains a behavioral attachment to the mother). He also suggested that metabolic weaning is more likely a gradual process occurring over time and more likely to occur in March-April, preceding the next reproductive season. In many otariids, the length of the lactation period varies among individuals and 'weaning' occurs over a period of time, rather than at a single point of time as with phocids (Lee *et al.*, 1991). Using an allometric relationship between weaning mass and maternal mass for otariids (Kovacs and Lavigne, 1992), and assuming a maternal mass of 530 lbs. (240 kg) (midpoint of range of maternal masses, 386.8 - 663 lbs (175 - 300 kg) (Calkins and Pitcher 1982), Steller sea lions could be expected to wean when achieving a body mass of 159 - 183 lbs (72 - 83 kg). According to growth data of Calkins and Pitcher (1982), this is achieved at an age of 11 months, and assuming a median pupping date of June 12, is an age reached in mid-May. A weaning age of 11 months was also used in analyses of comparative mammalian weaning by Lee *et al.* (1991). The transition to nutritional independence may, therefore, occur over a period of months as the pup begins to develop essential foraging skills, and depends less and less on the adult female. The length of the nursing period may also vary as a function of the condition of the adult female. The nature and timing of weaning is important because it determines the resources available to the pup during the more demanding winter season and, conversely, the demands placed on the mother during the same period. A bioenergetic model suggested that a 10 year old female nursing a pup in the spring had to consume twice as much energy as a same age female without a pup (Winship, 2000). The maintenance of the mother-offspring bond may also limit their distribution or the area used for foraging (see Section 3.1.6).

Relatively little is known about the life history of sea lions during the juvenile years between weaning and maturity. Female growth is asymptotic, and reaches 87% of the asymptote during their third year (Winship *et al.*, 2001), a size typically associated with puberty in female pinnipeds (Laws, 1956). Pitcher and Calkins (1981) found that females reach sexual maturity between 2-8 years of age, with an average age of first

pregnancy at 4.9 ± 1.2 years, and may breed into their early twenties. The available literature indicates an overall reproductive (birth) rate on the order of 55% - 70% or greater (Pike and Maxwell, 1958; Gentry, 1970; Pitcher and Calkins, 1981; Pitcher *et al.*, in press). York (1994) derived age-specific fecundity rates based on data from Calkins and Pitcher (1982). Those rates illustrate a number of important points and assumptions. First, the probability of pupping is rare (about 10%) for animals 4 years of age or younger. Second, maturation of 100% of a cohort of females occurs over a prolonged period which may be as long as 4 years (starting at age 3 or 4). Third, the reported constancy of fecundity extending from age 6 to 30 indicates that either senescence has no effect on fecundity, or our information on fecundity rates is not sufficiently detailed to allow confident estimation of age-specific rates for animals older than age 6. Given the small size of the sample taken, the latter is a more likely explanation for such constancy.

For mature females, the reproductive cycle includes mating, gestation, parturition, and nursing or post-natal care. Mating occurs about one to 2 weeks after giving birth (Gentry 1970). Copulation may occur in the water, but mostly occurs on land (Pitcher and Calkins, 1998; Gentry, 1970; Gisiner, 1985). The gestation period is probably about 50 to 51 weeks, but implantation of the blastocyst is delayed until late September or early October (Pitcher and Calkins, 1981). Due to delayed implantation, the metabolic demands of a developing fetus are not imposed on the female until well into fall and early winter. After parturition (birth), females nurse their pups over a period of months to several years. The reproductive success of an adult female is determined by a number of factors within a cycle and over time through multiple cycles. The adult female's ability to complete this cycle successfully is largely dependent on the resources available to her. While much of the effort to explain the Steller sea lion decline has focused on juvenile survival rates, considerable evidence suggests that decreased reproductive success may also have contributed to the decline.

- Young females collected in the 1970s were larger than females of the same age collected in the 1980s (Calkins *et al.*, 1998). As size, as well as age, may influence the onset of maturity, females in the 1980s would also be more likely to mature and begin to contribute to population productivity at a later age.
- Pitcher *et al.* (1998) provide data from the 1970s and 1980s that suggest a high pregnancy rate after the mating season (97%; both periods), which declined to 67% for females collected in the 1970s and 55% for females collected in the 1980s. These changes in pregnancy rate suggest a high rate of fetal mortality that could be a common feature of the Steller sea lion reproductive strategy (i.e., may occur even when conditions are favorable and population growth is occurring), but is more likely an indication of stress (possibly nutritional) experienced by individual females.
- The observed differences in late pregnancy rates (67% in the 1970s and 55% in the 1980s) were not statistically significant. However, the direction of the difference is consistent with the hypothesis that reproductive effort in the 1980s was compromised.
- Pitcher *et al.* (1998) did observe a statistical difference in the late season pregnancy rates of lactating females in the 1970s (63%) versus lactating females in the 1980s (30%). This difference indicates that in contrast to lactating females in the 1970s, lactating females in the 1980s were less able to support a fetus and successfully complete consecutive pregnancies.

Male growth is also asymptotic, but constant until about year 6 and thus males grow at a greater rate for a longer period than do females (Winship *et al.*, 2001). Males also reach sexual maturity at about 3 - 8 years old, but do not have the physical size or skill to obtain and keep a breeding territory until they are nine years of age or older (Pitcher and Calkins, 1981). A sample of 185 territorial males from Marmot, Atkins,

Ugamak, Jude, and Chowiet Islands in 1959 included animals 6 - 17 years of age, with 90% from 9 - 13 years old (Thorsteinson and Lensink, 1962). Males may return to the same territory for up to 7 years, but most return for no more than 3 years (Gisiner, 1985). During the breeding season, males may not eat for 1 to 2 months. The rigors of fighting to obtain and hold a territory and the physiological stress of the mating season reduces their life expectancy. Males rarely live beyond their mid-teens, while females may live as long as 30 years.

3.1.1.5 Survival

Much of the recent effort to understand the decline of Steller sea lions has been focused on juvenile survival, or has assumed that the most likely proximate explanation is a decrease in juvenile survival rates. This contention is consistent with direct observations and a modeling study, and is consistent with the notion that juvenile animals are less adept at avoiding predators and obtaining sufficient prey for growth and survival.

The direct observations consist of low resighting rates at Marmot Island of 800 pups tagged and branded at that site in 1987 and 1988 (Chumbley *et al.*, 1997) and observations of relatively few juveniles at Ugamak Island (Merrick *et al.*, 1988). The low resighting rates do not themselves confirm that the problem was a corresponding drop in juvenile survival, but only that many of the marked animals were lost to the Marmot Island population. Migration to other sites where they were not observed is a possibility, but unlikely given the observations of relatively high site fidelity of animals returning to breed at their natal site. If the “loss” of these animals is viewed in the context of the overall sea lion decline in the central GOA (from 1976 to 1994 the number of non-pups counted at Marmot Island declined by 88.9% and by 76.9% at the 14 other trend sites in the Gulf; Chumbley *et al.*, 1997), then a significant increase in juvenile mortality is a much more plausible conclusion.

Modeling by York (1994) suggests that the observed decline in sea lion abundance in the GOA may have been due to an increase in juvenile mortality. York used the estimated rate of decline between the 1970s and the 1980s, and the observed shift in the mean age of adult females (3 years of age) to explore the effects of changes in adult reproduction, adult survival, and juvenile survival. While she pointed out that the observed decline did not rule out all other possible explanations, she concluded that the observed decline is most consistent with a decrease in juvenile survival on the order of 10% - 20% annually.

However, juvenile survival is not assumed to be the only factor influencing the decline of the western population of Steller sea lions. Evidence indicating a decline in reproduction was presented in the previous section. In addition, changes in adult survival may also have contributed to the decline. At present, survival rates for adult animals can not be determined with sufficient resolution to determine if those rates have changed over time or are somehow compromised to the extent that population growth and recovery are compromised.

3.1.1.6 Age distribution

Two life tables have been published with age-specific rates. The first was from Calkins and Pitcher (1982) and was based on sea lions killed in the mid to late 1970s. York (1994) created a second life table using a Weibull model from the same data from Calkins and Pitcher (1982). A comparison of data collected between the 1970s and the 1980s (Calkins and Goodwin 1988) showed a 1.55 year increase in the mean age of reproductive females.

Both samples of sea lions were from the same area but collected over different time periods. Independent of the samples were observations of the rate of decline of the population. These three pieces of information were combined (York 1994) to suggest that increased juvenile mortality may have been an important proximate factor in the decline of Steller sea lions. That is, such a shift in mean age would occur as the adult population aged without expected replacement by recruitment of young females.

3.1.1.7 Prey and Foraging Behavior

3.1.1.7.1 Methodology and potential biases

Historically, diet studies on marine mammals were based on the remains of prey in the stomach contents of the predator. Currently, the primary method of identifying prey species consumed by pinnipeds is through analysis of bony remains in fecal (scat) collections. The interpretation of predator diet through the use of scat was first developed for terrestrial studies and has been adapted for use in marine mammal trophic studies over the past two decades. Scat is a reliable tool for monitoring seasonal and temporal trends in predator diets without the need to euthanize the animal.

Typically, the rank importance of any given prey species in marine mammal diet studies is based on some combination of two factors: the number of individuals of a particular species represented across all samples (prey number); and the number of samples containing that species across all samples containing prey remains (frequency of occurrence). All methods of diet evaluation (stomachs, lavage, regurgitations, scat, enema, fatty acids, and stable isotopes analyses) in marine mammals have their own set of biases that variably affect estimates of prey volume, weight, number, rank and frequency of occurrence (Fitch and Brownell, 1968; Perrin *et al.*, 1973; Jobling and Breiby, 1986). For instance, stomach contents from an individual animal may represent an accumulation of a number of meals over an extended period of time since certain prey parts such as squid beaks or large fish bones get trapped in stomach folds where they digest very slowly, or accumulate until regurgitated. An accumulation of prey parts predictably overestimate the importance of some prey types over others. Regurgitations (spewings) represent a very small portion of the overall diet and primarily that of the largest prey items consumed. Scat remains by comparison typically represent meals eaten 12 - 72 hours prior and tend to underestimate the size of prey consumed since small items pass through the digestive tract more readily (and with less erosion) than large items. Accordingly, diet studies should be interpreted with consideration of the method used to collect prey samples.

Steller sea lions eat a broad range of prey that vary in adult body size from approximately 10-80 cm in body length. The most recent diet study of the western stock (Sinclair and Zeppelin, submitted) indicates that prey remains in scat are primarily from late stage juvenile to adult size fish. However, these estimates are qualitatively based on visual comparison with museum reference specimens. To date, estimates of prey size have remained qualitative due to the limited number of intact otoliths (fish earbones) recovered from Steller scats. Otoliths recovered from pinniped stomachs and scats are typically used to estimate size of fish prey using otolith to body length regressions. However, relatively few otoliths are recovered in Steller sea lion scats and those that are found are usually highly degraded due to erosion in the stomach. Other skeletal remains are found in Steller sea lion scat in greater abundance and in better condition than otoliths, but until recently (Zeppelin *et al.*, in prep), no technique existed to quantify bone size to fish length.

Zeppelin *et al.* (in prep) developed regressions to estimate fish length using six diagnostic bones (other than otoliths) from Pacific cod (*Gadus macrocephalus*), walleye pollock (*Theragra chalcogramma*), and Atka mackerel (*Pleurogrammus monopterygius*). For all species and all elements, regressions showed a high

degree of correlation between the size of the element and the fork length of prey ($r^2 \geq 0.85$). Fork length of prey consumed by Steller sea lions was predicted by applying these regression models to skeletal structures recovered from scats collected during 1998 and 1999 across the range of the western stock, confirming qualitative estimates that Steller sea lions largely consume adult fish (Sinclair and Zeppelin, submitted). These regressions will eventually allow quantitative comparisons between the size of prey consumed by Steller sea lions and the commercial fish catch in Bering Sea and Gulf of Alaska.

3.1.1.7.2 Foraging Distribution and Depths

At least three types of telemetry are (or have been) used to study sea lion foraging. Very high frequency (VHF) telemetry can be used to determine presence or absence of an animal and, to some extent, animal location and whether it is on land or in the water. The use of VHF telemetry to determine the presence or absence of an animal can be used to infer the occurrence and length of foraging trips (e.g., Brandon, 2000; Merrick and Loughlin, 1997), and movement patterns between sites that can be monitored manually, remotely, or automatically by VHF receivers.

Satellite-linked telemetry is being used to determine animal location and, when coupled with time-depth recorders, diving patterns (e.g., Merrick *et al.*, 1994). Satellite-linked telemetry provides an opportunity to collect information on animal location without having to recapture the animal to collect stored data. At present, satellite-linked telemetry is the most cost-effective means of assessing the distribution of foraging animals and thereby determining those regions that are critical for Steller sea lions.

Stomach telemetry is being developed and offers an opportunity to determine when an animal has consumed prey, rather than requiring the investigator to infer feeding from diving behavior. Stomach telemetry, in combination with satellite-linked telemetry, may provide greater understanding of foraging behavior and discrimination of at-sea activities that may or may not be related to foraging (Andrews, 1998).

Steller sea lion foraging distribution is based on sightings at sea or observations of foraging behavior (or presumed foraging behavior) in areas such as the southeastern Bering Sea (Fiscus and Baines, 1966; Kajimura and Loughlin, 1988; NMFS unpublished data¹ from the Platform-of-Opportunity Program [POP]), records of incidental take in fisheries (Perez and Loughlin, 1991), and satellite telemetry studies (e.g., Merrick *et al.*, 1994; Merrick and Loughlin, 1997). Observations and incidental take of sea lions (Loughlin and Nelson, 1986; Perez and Loughlin, 1991) in the vicinity of Seguam Pass, the southeastern Bering Sea, and Shelikof Strait provided a basis for establishment of those areas as critical habitat.

The POP database provides our best overall view of the foraging range or distribution of Steller sea lions in the Bering Sea and the western/central Gulf of Alaska (Figure 3.1-6). This database and the locations of sea lions taken incidentally in groundfish fisheries (1973 - 1988, Perez and Loughlin, 1991), indicate that sea lions disperse widely to forage throughout much of the Bering Sea and the GOA, at least as far out as the continental shelf break. Such broad dispersal may be essential to sea lion populations to take advantage of distant food resources and, as a consequence, limit intra-specific competition near rookeries and haulout sites. However, this database should be viewed with some caution. The sightings in the POP database were collected over a period of 4 decades and do not reflect any natural changes that may have occurred in sea lion foraging patterns during that period. Similarly, foraging range, as indicated by such sightings, would be expected to change over time due to the severe decline of the species in the last two decades. In addition,

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the database is biased as a reflection of overall foraging dispersion by the location of sighting effort. That is, a sighting at a particular location indicates sea lion presence at that site, but the lack of sightings at a site could mean that the site is not important for foraging or it could mean that there was insufficient sighting effort in that area. Also, it is not clear that each sighting represents a different animal, and it is possible that some sightings were of the same animal. Furthermore, the area may be important to sea lions during times of the year not covered intensively by sightings. Finally, the sighting database does not include information on the age and sex of the sighted animal.

Telemetry studies suggest that foraging distributions vary by individual, size or age, season, site, and reproductive status (Merrick and Loughlin, 1997). The foraging patterns of adult females differed during summer months when females were with pups versus winter periods when considerable individual variation was observed, but may be attributable to the lactation condition of the females. Trip duration for females ($n = 14$) in summer was approximately 18 - 25 hours. For five of those females that could be tracked, trip length averaged 10.5 miles (17 km) and they dove approximately 4.7 hours per day. For five females tracked in winter months, mean trip duration was 204 hours, mean trip length was 82.5 miles (133 km), and they dove 5.3 hours per day. The patterns exhibited by females in winter varied considerably, from which the investigators inferred that two of them may still have been supporting a pup. Those two females continued to make relatively shorter trips (mean of 32.9 miles [53 km] over 18 hours) and dove 8.1 hours per day, whereas the other three ranged further, dove 3.5 hours per day, and spent up to 24 days at sea. Five winter young-of-the-year exhibited foraging patterns intermediate between summer and winter females in trip distance (mean of 18.6 miles [30 km]), but shorter in duration (mean of 15 hours), and with less effort devoted to diving (mean of 1.9 hours per day). Estimated home ranges (mean \pm 1 SE) were 197 ± 38.4 miles² (319 ± 61.9 km²) for adult females in summer, $29,499 \pm 16,556.4$ miles² ($47,579 \pm 26,704$ km²) for adult females in winter, and $5,701.5 \pm 4,215.4$ miles² ($9,196 \pm 6,799$ km²) for winter young-of-the-year. The sea lions used in Merrick and Loughlin's (1997) study were from the GOA (Sugarloaf Island, Latax Rocks, Marmot Island, Long Island, Chirikof Island, Atkins Island, and Pinnacle Rock), and the BSAI region (Ugamak Island and Akun Island). This information is, therefore, directly pertinent to the action areas for both the GOA and BSAI fisheries, although it is perhaps most relevant to the GOA action area.

Overall, the available data suggest two types of foraging patterns: 1) foraging around rookeries and haulout sites that is crucial for adult females with pups, pups, and juveniles, and 2) foraging that may occur over much larger areas where these and other animals may range to find the optimal foraging conditions once they are no longer tied to rookeries and haulout sites for reproductive or survival purposes.

The sea lions in the Merrick and Loughlin (1997) study tended to make relatively shallow dives, with few dives recorded at greater than 820 feet (250 m) (Figure 3.1-7). Maximum depth recorded for each of the five summer adult females was in the range from 328 to 820 feet (100 to 250 m), and maximum depth for the five winter adult females was greater than 820 feet. The maximum depth measured for winter young-of-the-year was 236.2 feet (72 m). These results suggest that sea lions are generally shallow divers, but are capable of deeper dives (i.e., greater than 820 feet). The winter young-of-the-year sea lions were instrumented from November to March, when they were about 5 - 9 nine months old and may have still been nursing. At this age, they are just beginning to develop foraging skills, which may take years to learn. The diving depths and patterns exhibited by these young-of-the-year are not indicators of the foraging patterns of older juveniles (1 - 3-year-olds). For example, Swain and Calkins (1997) report dives of a 2-year-old male sea lion to 826.6 feet (252 m), and regular dives of this animal and a yearling female to 492 - 820 feet (150 m - 250 m). Clearly, if young-of-the-year are limited to relatively shallow depths, and older animals are capable of diving to much greater depths, then those younger animals are just beginning to develop the diving and foraging skills necessary to survive. The rate at which they develop those skills and begin to dive to greater depths

or take prey at greater depths is unknown, but probably occurs rapidly after weaning to take advantage of otherwise unavailable prey resources. The ADF&G is currently studying the ontogeny of dive behavior in young Steller sea lions.

A recent study analyzed data from 13 pup and 12 yearling Steller sea lions equipped with satellite dive recorders in the Gulf of Alaska/Aleutian Islands ($n = 18$), and Washington ($n = 7$) from 1994-2000 (Loughlin *et al.*, unpublished). A total of 1413 days of transmission ($\bar{x} = 56.5$ days, range 14.5-104.1 days) were received. They recorded 222,073 dives, which had a mean depth of 60.4 feet (18.4 m) (range of means 19.0 - 222.7 feet [5.8-67.9 m]). Alaskan pups dove briefer and shallower (mean depth $\bar{x} = 25.3$ feet [7.7 m], mean duration $\bar{x} = 0.8$ min, mean maximum depth $\bar{x} = 84.3$ feet [25.7 m], and maximum depth $\bar{X} = 826.6$ feet [252 m]) than Alaskan yearlings ($\bar{x} = 54.4$ feet [16.6 m], $\bar{x} = 1.1$ minutes, $\bar{x} = 63.4$ m, 944.6 feet [288 m], respectively), with Washington yearlings diving the longest and deepest ($\bar{x} = 129.2$ feet [39.4 m], $\bar{x} = 1.8$ minutes, $\bar{x} = 474.0$ feet [144.5 m], and 1,075.8 feet [328 m], respectively).

Loughlin *et al.*, unpublished) described three types of movements for these young sea lions, long-range trips (greater than 8 miles and greater than 20 hours), short-range trips (less than 8 miles and less than 20 hours), and transits to other sites. Transits began as early as 7 months of age, occurred more often after 9 months of age and ranged between 3.5 - 245 miles. Long-range trips started around 9 months of age and occurred most frequently at around the time of weaning while short-range trips happened almost daily (.9 trips/day, $n = 426$ trips).

3.1.1.7.3 Prey Species and Size

Steller Sea Lion Diet, Western Stock - Current²

The most recent analysis of Steller sea lion diet compares trends in prey species consumption between summer and winter, when juveniles are first learning to forage on their own. (Sinclair and Zeppelin, submitted). Steller sea lion scats were collected (1990-1998) from 31 rookeries (May-September) and 31 haulout sites (December-April) across the U.S. range of the western stock resulting in a sample of 3,762 scats with identifiable prey remains. As is typical in marine mammal diet studies prey remains were identified to the lowest possible taxon using museum reference specimens. The relative importance of each prey species was based on their frequency of occurrence (FO).

Frequency of occurrence values combined across years, seasons, and sites depicted walleye pollock (*Theragra chalcogramma*) and Atka mackerel (*Pleurogrammus monopterygius*) as the two dominant prey species, followed by Pacific salmon (Salmonidae) and Pacific cod (*Gadus macrocephalus*). Other primary prey species consistently occurring at frequencies of 5% or greater included arrowtooth flounder (*Atheresthes stomias*), Pacific herring (*Clupea pallasii*), Pacific sand lance (*Ammodytes hexapterus*), Irish lord (*Hemilepidotus sp.*), and cephalopods (squid and octopus). Species that occurred among the top three prey items on certain islands included: snailfish (Liparididae), rock greenling (*Hexagrammos lagocephalus*), kelp greenling (*Hexagrammos decagrammus*), sandfish (*Trichodon trichodon*), rock sole (*Lepidopsetta bilineata*), northern smoothtongue (*Leuroglossus schmidtii*), skate (Rajidae), and smelt (Osmeridae) (Figure 3.1-8).

²Discussion of current diet trends in the western stock of Steller sea lions is based on a recently submitted draft publication (Sinclair and Zeppelin, submitted) and should not be cited without permission from the authors.

Sites where the FO were most similar were identified using Principal Components and Agglomerative Hierarchical Cluster Analysis (Ward, 1963; Ramsey and Schafer, 1996) resulting in regions of diet similarity. These newly defined diet regions were used to compare regional and seasonal differences in prey. The diet divisions closely paralleled those defined as metapopulations based on patterns in population decline by York *et al.* (1996) suggesting that diet and decline are linked (Figure 3.1-9).

Chi-square analysis demonstrated significantly ($P = 0.01$) strong seasonal patterns in diet within each of the defined diet regions (island groupings as defined by cluster analysis). Pacific cod FO was significantly larger in winter in every region. Salmon FO was significantly lower during winter in the western Gulf of Alaska through the eastern Aleutian Islands, and higher in winter throughout the central and western Aleutian Islands. In the western Gulf, where arrowtooth flounder is most abundant in scats and well represented year-round, its FO was significantly lower in winter. Atka mackerel was significantly lower in the winter in the central and western Aleutians where it is the dominant prey species year-round. Forage fishes (herring and Pacific sand lance) are significantly different between seasons, however, there is no general trend among the regions. Walleye pollock is an important prey year-round in all regions up to the central Aleutian Islands where it is replaced by Atka mackerel. Likewise, cephalopod FO was not significantly different between seasons in any Region. Irish lord FO was generally higher in winter than in summer and though rarely occurring during summer and not included in Chi-square analysis, sandfish and snailfish have relatively high occurrences during the winter across all regions.

Diet diversity, calculated using Shannon's index of diversity (Ludwig and Reynolds, 1988), indicated that the Unimak Pass area as well as Sea Lion Rock (Amak Island) on the continental shelf just eastward of the pass encompassed the regions of highest prey diversity in this study. In the midst of precipitous population declines range wide among the western stock (Loughlin *et al.*, 1992), Amak Island was among 5 other rookeries identified by York *et al.* (1996) that demonstrated persistently stable or increasing population counts: Amak, Akun, Akutan, Chernabura, Clubbing, Ugamak. The York *et al.* (1996) temporal model for extinction of the western stock predicted that in the face of extinction of all other sites, these six would remain viable. All of these sites fall within Regions 2 and 3 as defined in this study, regions of highest diversity and greatest overlap in prey matrices between regions in this study (Figure 3.1-9). Implications of the importance of diversity in otariid diet (Merrick *et al.*, 1997; Sinclair *et al.*, 1994), though difficult to measure, should be further addressed, with special attention given to the dynamics of physical and bottom-up processes that influence nearshore habitat of rookery regions and ultimately, the population stability of Steller sea lions.

Based on the patterns in prey consumption presented in this (Sinclair and Zeppelin, submitted) and earlier studies (Fiscus and Baines, 1966; Pitcher, 1981; Calkins, 1998) Steller sea lions specialize feeding throughout the water column in the epipelagic (herring), demersal (arrowtooth flounder), and semi-demersal (pollock, Atka mackerel) zones. While the size of prey consumed undoubtedly varies with the age and sex of sea lion sampled, the remains of primary prey represented in this study are largely from adult fish (Zeppelin *et al.*, in prep). The seasonal and regional patterns in prey consumption by Steller sea lions presented in this study, along with known distributions of their primary prey, indicate that Steller sea lions target prey when they are densely schooled in spawning aggregation nearshore (over or near the continental shelf) or along oceanographic boundary zones. This is true in summer when collected scats are primarily from adult females, and in winter when scats are presumably from some increased proportion of juveniles and adult males as well as females.

The close parallel of these data (Sinclair and Zeppelin, submitted) with those of metapopulation patterns of decline (York *et al.*, 1996) suggests that diet and decline of Steller sea lions is linked; that diet diversity is

highest where population trends are most positive; and that regional diet patterns generally reflect regional foraging strategies learned at or near the natal rookery site on seasonally dense prey patches characteristic of that area. These data do not reflect Steller sea lion diet outside the range of the U.S. western stock. Goto and Shimazaki (1998) examined stomachs from 67 Steller sea lions killed by hunters off the coast of Hokkaido, Japan during 1994-1996. The most common prey was walleye pollock, but other common prey included Pacific cod, saffron cod, cephalopods, and flatfishes (Pleuronectidae).

Steller sea lion diet, western stock - historical

In terms of the species of fish eaten by Steller sea lions, recent diet work (Sinclair and Zeppelin, submitted) compares most closely with studies conducted since the mid-1970s. In studies conducted along the range of the western stock between 1958 and 1969, pollock were completely absent from Steller diet (Mathisen *et al.*, 1962; Thorsteinson and Lensink, 1962; Tikhomirov, 1964; Fiscus and Baines, 1966). The high occurrence of pollock in Sinclair and Zeppelin (submitted) this study is comparable to diet studies conducted since 1975 (Calkins, 1998; Frost and Lowry, 1986; Merrick *et al.*, 1997³; Pitcher, 1981) and possibly prior to the 1950s when Imler and Sarber (1947) reported pollock in 2 stomachs collected near Kodiak Island in 1945-1946. Sinclair and Zeppelin (submitted) also highlight the importance of Pacific cod in Steller diet during the winter months. Prior to this work, relatively few papers have focused on winter diet, so it is difficult to assess whether consumption of Pacific cod by Steller sea lions is a recent trend. Pacific cod was a top prey item in Calkins (1998) Bering Sea winter collections, and in stomachs collected in winter in the Gulf of Alaska 1973-1975 (Pitcher 1981). Overall, the most common prey items in studies prior to the mid-1970s included: capelin (*Mallotus villosus*), sand lance, cephalopods, herring, greenlings (Hexagrammidae), rockfishes, and smelts. Capelin, which were important in Steller diet through the 1970s (Fiscus and Baines, 1966; Pitcher, 1981) do not have an occurrence greater than 5% in recent studies. Salmon were present in early studies, but, not at the frequencies found across the western range during the summer. The occurrence of flatfish, especially Arrowtooth flounder, in the Gulf of Alaska is substantially higher now than any previous studies. Cephalopods were among the top prey items found in Steller sea lion stomachs in many early studies (Mathisen *et al.*, 1962; Pitcher, 1981; Thorsteinson and Lensink, 1962), sometimes ranking as the most frequently occurring prey item (Fiscus and Baines, 1966). Cephalopod occurrence in Sinclair and Zeppelin (submitted) was primarily limited to the central and western Aleutian Islands and highest during the summer months, but never reached the high frequencies of the 1960s. The difference in cephalopod values between recent scat and historical stomach based diet studies may be due to differences in representation of cephalopod beaks in scats versus stomachs.

3.1.1.7.4 Prey Quality

An important consideration in evaluating effects of changing diets or prey abundance on Steller sea lions is the quality of the prey. Lipid content, and therefore energy density, varies greatly among Steller sea lion prey species, and within prey species depending upon life history stage, location and time of year (Stansby, 1976; Van Pelt *et al.*, 1997; Payne *et al.*, 1999; Anthony *et al.*, 2000). Atka mackerel and gadids are generally lower energy dense prey species (ranging within about 3 kJ/g - 6 kJ/g, though few data exist for Atka mackerel), while forage fish such as eulachon, herring, or capelin have generally higher energy contents (up to about 11 kJ/g). Because energy densities are seasonally variable, this is not an absolute relationship. For

³Merrick *et al.* 1997 was based on portions of the 1990-1993 dataset incorporated into Sinclair and Zeppelin (submitted).

example, capelin and sandlance declined in lipid content, and therefore energy density, throughout the summer, from 6.7 kJ/g to 3.7 kJ/g and 6.5 kJ/g to 4.8 kJ/g respectively (Anthony *et al.*, 2000).

In addition to considerations of prey energy content, vitamins and other metabolites are essential for adequate nutrition. There is evidence from captivity that marine mammal diets entirely or largely comprised of clupeid (herring) and osmerid (smelt) induces thiamin deficiencies (Geraci 1981). This has not been diagnosed with certainty in otariids, though it was thought to be observed in California sea lions. The potential for fish-induced anemia in marine mammals was also highlighted by Geraci (1981) based on symptoms expressed in terrestrial mammals fed primarily gadoid fish. This condition was found in mink fed thawed raw gadoid fish, but not fresh raw fish, and has not been directly observed in marine mammals. However, Thompson *et al.* (1997) and Fadely (1997) referenced this mechanism when circumstantially associating similar hematologic findings in free-ranging harbor seals with evidence for switching between clupeid and gadoid diets.

The ultimate net energy gain imparted to an animal from ingesting a particular prey item not only depends upon the energy content of the prey, but also on the costs associated with traveling to, finding, capturing, handling, and digesting the prey. It thus also depends on the prey item's individual size, total biomass, availability, behavior, degree of aggregation, temporal and spatial distribution, and so on. That is, the value of any particular prey type depends on the net gain to a sea lion from foraging on that prey, and net gain is a function of multiple factors of which lipid content is an important, but not the only, determinant.

3.1.1.7.5 Foraging - Integration and Synthesis

Foraging patterns of Steller sea lions are still far from being completely described. However, the available information suggests that:

- Steller sea lions are land-based predators but their attachment to land and foraging patterns/distribution varies considerably as a function of age, sex, site, season, and reproductive status, and as a function of prey availability and environmental conditions.
- Steller sea lions tend to be relatively shallow divers but are capable of (and apparently do) exploit deeper waters (e.g., to beyond the shelf break).
- Foraging sites relatively close to rookeries may be particularly important during the reproductive season when lactating females are limited by the nutritional requirements of their pups.
- Pups dependent upon mothers for nutrition tend not to disperse greatly and remain relatively nearshore conducting shallow dives.
- Yearlings that have reached nutritional independence greatly increase their foraging area, and begin deeper diving.
- Food availability may be extremely important during April - June, when pups are likely to be making a transition to nutritional independence, and the energy requirements of pregnant females are about double that of nonpregnant females.

- Dominant prey items vary with region and season, but pollock, Atka mackerel, Pacific cod and salmon are generally the most common or dominant prey.
- Steller sea lions consume a variety of demersal, semi-demersal, and pelagic prey, indicating a potentially broad spectrum of foraging styles.
- Diet diversity may influence status and growth of Steller sea lion populations.
- The life history and spatial/temporal distribution of important prey species are likely important determinants of sea lion foraging success
- The broad distribution of sea lions sighted in the POP database indicates that sea lions also forage at sites distant from rookeries and haulout sites.
- The availability of prey at these sites may be crucial in that they allow sea lions to take advantage of distant food sources, thereby mitigating the potential for intraspecific competition for prey in the vicinity of rookeries and haulout sites.
- The question of whether competition exists between the Steller sea lion and BSAI or GOA groundfish fisheries is a question of sea lion foraging success. For a foraging sea lion, the net gain in energy and nutrients is determined, in part, by the availability of prey or prey patches it encounters within its foraging distribution. Competition occurs if the fisheries reduce the availability of prey to the extent that sea lion condition, growth, reproduction, or survival are diminished, and population recovery is impeded.

3.1.1.8 Physiology and Nutrition

Fundamental to an evaluation of the effects of commercial fisheries on Steller sea lions is an understanding of the physiological adaptations and nutritional requirements that underlie the sea lion's role in the Bering Sea and Gulf of Alaska ecosystems. Steller sea lions spend time on land at rookeries and haulout sites for reproduction, lactation, molting and resting, and undertake foraging trips to sea. The relative time spent at sea depends upon age and size, season, reproductive status, and the availability of forage (Boyd, 1995; Boyd, 1996). Trips to sea may be made for a variety of reasons, for foraging but also for seasonal or age-specific movements along the coast. While foraging, swimming and diving behavior are controlled by a compromise between the necessity to breathe at the surface and to submerge to seek and consume prey. In addition to the abundance and distribution of prey, the time a sea lion spends submerged will depend upon physiological adaptations for maximizing time underwater. This will be a result of the how fast oxygen stores are utilized (that is, metabolic rate), and how much oxygen is stored in the body, and the conflicting demands of diving and exercise (Castellini, 1991; Boyd, 1997). Pinnipeds exhibit many physiological strategies to increase dive duration (Boyd and Croxall, 1996; Boyd, 1997).

During a dive, approximately 47% of a sea lions oxygen stores are in blood, with 35% in muscle and the remainder in the lungs (Kooyman, 1985). There is a considerable developmental component until the oxygen storage ability of an otariid is fully matured (Horning and Trillmich, 1997), because of increases in blood volume, muscle myoglobin and body mass. Likewise, juveniles operate at metabolically higher rates than adults (Lavigne *et al.*, 1986; Costa, 1993). Thus, younger sea lions do not have the same capacity to stay

submerged (and hence dive to as great of depths) as adults, which consequently affects their ability to acquire prey and thus choice of foraging strategies.

Sea lions have a streamlined shape that minimizes the cost of transport while swimming (Feldkamp, 1987; Stelle *et al.*, 2000). Laboratory measurements of swim speed and drag in 3 year old Steller sea lions showed that they preferred to swim at a mean velocity of 3.41 m s^{-1} ($2.9\text{-}3.4 \text{ m s}^{-1}$), equivalent to 1.46 body lengths per second (Stelle *et al.*, 2000), a speed found to be the minimum cost of transport for California sea lions (Feldkamp, 1987). Williams *et al.* (1991) found that the average ventilation time did not change with swim speed in California sea lions. That is, time spent submerged did not change as swim speeds increased to 4.37 yd s^{-1} (4 m s^{-1}). Because of anatomical adaptations, sea lions appear to require shorter times for lung tidal volume exchange than do seals (Williams *et al.*, 1991). Skeletal muscles of pinnipeds (and sea lions) are adapted for aerobic metabolism of lipids during hypoxic conditions of diving and exercise (Kanatous *et al.*, 1999). Lipid stores in swimming muscle were sufficient to meet the resting muscle metabolism for 17 hours in Steller sea lions (Kanatous *et al.*, 1999).

Food Intake Requirements

Kastelein *et al.* (1990) measured the daily food intake rate of 10 captive Steller sea lions for a 15 year period. The sea lions were fed 3 - 6 times per day on a mixture that averaged 22% mackerel, 40% herring, 10% sprat, 14% whiting (a gadid), and 14% squid based on weight. They found large seasonal variation in daily intake rates, with a particularly strong depression of consumption during summer months in males between 8-15 years old. Seasonality of intake was presumably related to hormonal influences, changes in energy requirements due to reproduction, and seasonal temperature changes. Much of these effects could be influenced or transduced by changes in sea lion body mass or composition, which they could not measure.

To estimate the amount of food required by Steller sea lions in the wild, more detailed measurements of metabolic rates and food intake requirements have been measured in captivity. The amount of energy derived from food can be described by following the bioenergetic scheme which details the efficiencies and costs of metabolizing food at various steps in the digestive process (Lavigne *et al.*, 1982). Proportions of proteins, lipids, or undigestible matter in a prey item affect its ability to be absorbed and digested in the body. Energy available in the prey is lost throughout the digestive process before it becomes available for activity, growth or reproduction by the sea lion, and varies depending upon the relative amounts of protein and lipid or energy density in the diet item (Rosen and Trites, 2000a; Rosen and Trites, 2000b). Laboratory measurements of nutritional requirements seek to control multiple sources of variation that affect the amount of energy derived from prey. Then, if the costs of various activities are also determined, it is possible to model the nutritional requirements of sea lions in the wild. This provides a basis to determine the effects of consuming prey items with varying energy content.

Assimilation (dry-matter) efficiencies of northern fur seals, California sea lions, and Steller sea lions fed herring range from 86.9% to 93% (Miller, 1978; Fadely *et al.*, 1990; Fadely *et al.*, 1994; Rosen *et al.*, 2000; Rosen and Trites, 2000a), and for pollock between 81.3% - 90% (Miller, 1978; Fadely *et al.*, 1994; Rosen and Trites, 2000a). In terrestrial mammals, larger meal sizes or increased feeding frequency may decrease assimilation efficiencies (Golley *et al.*, 1965), but while these effects may have been observed in ringed seals (Parsons, 1977), harp seals (Keiver *et al.*, 1984) and northern fur seals (Fadely *et al.*, 1990), it was not observed in Steller sea lions (Rosen *et al.*, 2000) or in other studies of harp, ringed or grey seals (Ronald *et al.*, 1984; Lawson *et al.*, 1997a; Lawson *et al.*, 1997b). Because fish have relatively small indigestible fractions, there is a linear relationship between digestive efficiency and assimilation efficiency, and digestive

efficiencies are likewise high (Rosen *et al.*, 2000) but not equivalent because of prey composition. When compared to other pinnipeds, assimilative and digestive efficiencies of Steller sea lions are comparable (Rosen and Trites, 2000a).

However, the cost of consuming low-energy prey items may be greater than expected purely based on differences in prey energy content. Captive feeding trials using six Steller sea lions ranging in age between 0.9 and 4.5 years old were fed herring or pollock diets over periods of 7 - 24 days to measure differences in metabolism and digestive efficiency (Rosen and Trites, 1997; Rosen and Trites, 1999; Rosen and Trites, 2000a; Rosen and Trites, 2000b; Rosen *et al.*, 2000). Based on the gross energy content differences between the two diets, the captive sea lions were expected to consume about 35% - 65% more pollock than herring. However, results suggested that because of additional costs of digesting pollock and a slightly lower digestive efficiency relative to a herring diet, the sea lions would have had to consume about 35% - 80% more pollock than herring (Rosen and Trites, 2000b). The sea lions in that study were unable or unwilling to consume the additional amounts of pollock, and hence lost body mass in spite of also reducing resting metabolic rates (Rosen and Trites, 1999).

A Steller Sea Lion Recovery Team review of captive feeding studies performed to date suggested that these studies may not be generally representative of field situations (Didier 1999), a point that has also been highlighted by researchers conducting the studies (Fadely *et al.*, 1994; Rosen and Trites, 2000b). They cited the short duration, often less than two weeks, which may have been inadequate to trigger cues utilized by sea lions to adjust intake in response to dietary changes. Likewise, these studies did not directly measure changes in activity or body condition, which also affect food intake rates. As a goal of the captive feeding studies was specifically to detect differences in metabolism related to prey composition, sea lions were fed single-species diets to maximize potential treatment effects, and as such, these studies present important data regarding metabolic adjustments that can be made in response to diets of differing quality. However, the diet of free-ranging sea lions does not consist exclusively of single prey items, prey vary in energy density throughout the year, and it is also unknown how availability of prey may affect total intake (Rosen and Trites, 2000b).

A subsequent set of captive feeding studies are being conducted to address many of these concerns by performing feeding trials throughout the year, and by using mixed diets based on known diet compositions of free-ranging sea lions (Castellini, 2001). Using mixed diets based on pre-decline, decline, and Southeast Alaska prey frequencies of occurrence in sea lion diets, preliminary results indicate that in some seasons, body mass is defended regardless of diet, and sea lions adjust volume consumed; two sea lions showed relatively stable body composition regardless of diet, a third changed composition; and, all sea lions had maintained excellent health and condition regardless of diet. Thus, while a monospecific diet of low-fat prey may be nutritionally limiting to sea lions under certain conditions, results from feeding trials performed over longer periods with diet mixtures reflective of wild sea lions have the potential to refine our understanding of free-ranging dietary needs of sea lions. And though captive feeding studies can describe the metabolism of prey once ingested, they do not include components of foraging efficiency, or the cost to the sea lion of acquiring a certain prey type.

With estimates of food intake requirements, population size and age structure, and of activity, it is possible to generate estimates of food intake requirements for the entire population of Steller sea lions. Perez and McAllister (1993) estimated annual food consumption by the Steller sea lion population in the eastern Bering Sea as 18.5×10^4 mt, of which 14.1×10^4 mt (76 percent) was fish. Of the total annual fish consumption, commercial groundfish comprised 69 percent. Winship (2000) built a Steller sea lion bioenergetic model incorporating variability in estimates of sea lion growth, metabolism, activity, and diet to produce more

robust estimates of population food intake requirements. Estimates of annual prey biomass consumption varied seasonally, and by up to 12% among regions of Alaska due to differences in population size and diet composition (Winship, 2000). The Southeast Alaska population consumed the greatest biomass annually, while the central Aleutian Islands population consumed the least. Similar amounts of gadids were consumed annually by the Gulf of Alaska (7.64×10^4 mt) and Southeast Alaska populations (7.29×10^4 mt). The second largest single species consumption was of Atka mackerel by the central Aleutian Islands population (4.87×10^4 mt). Winship (2000) estimated that the total annual consumption of pollock by all sea lions was 6% of the total estimated pollock biomass attributed to natural mortality, and 19% of the total biomass removed by commercial fisheries. Steller sea lion predation accounted for a greater proportion (83%) of the estimated biomass of Atka mackerel annual natural mortality. However, this type of analysis does not consider spatial, temporal or local availability of prey to sea lions, particularly on scales relevant to foraging sea lions (Winship, 2000).

Physiology and Foraging - Integration

Field measurements of metabolic rate or energy consumption show that otariids generally operate at 3-6 times their basal metabolic rate while traveling and foraging (Costa *et al.*, 1989; Arnould *et al.*, 1996; and see Winship, 2000). This is higher than measurements for phocids, and reflects a high energy strategy for foraging. In general, otariids have adopted an “energy maximizer” type foraging strategy, which is characterized by high energy turnover. That is, sea lions expend comparatively (to phocids) high levels of energy to acquire relatively high levels of energy. This strategy is advantageous in highly productive ecosystems with concentrated and predictable prey (Costa, 1993).

Otariids can make adjustments to foraging strategies on many behavioral and metabolic scales. Changes in foraging trip duration and in time at a prey patch have been observed in response to prey availability (Boyd, 1996; Boyd, 1999; Andrews, 2001). Responses by sea lions will vary depending upon life history status, for example, whether an adult female is lactating or not, or whether a mother-pup pair is at a rookery (central place foraging), or foraging from multiple haulout sites (multiple central place foraging). This change in strategy is likely related to costs of lactation, when at some point it becomes more advantageous energetically for the female to move away from the rookery with the pup, though it is not yet weaned, to allow exploitation of prey with a higher rate of energy return (Boyd, 1998), either because of prey proximity, quality, or abundance at sites other than near the rookery.

Individual foraging strategies will vary depending upon prey location and quality. If prey are not shallow, travel costs increase to access the prey patch. At some combination of prey size, quality, number, catchability and depth, it will become suboptimal for a sea lion to forage on a given prey type (Boyd, 1997). This type of foraging decision was recently directly observed by Thomas and Thorne (2001), where sea lions in Prince William Sound were observed feeding on surface schooling herring, rather than diving to a deeper, though more concentrated, school of pollock.

3.1.1.9 Anthropogenic Sources of Mortality

Anthropogenic, or human-caused, sources of mortality can occur incidental to other actions, or through directed taking. Examples are mortalities that occur incidental to commercial fishing, through entanglement in derelict fishing gear or other debris, directly through subsistence harvests, or directly by illegal shooting or other action.

A primary source of data for the rate of mortalities that occur incidental to commercial groundfish fishing is from the North Pacific Groundfish Observer Program database. Based on recent observer data, minimum estimates of mean annual mortality for the BSAI groundfish trawl fishery are 7.0 (CV = 0.21) sea lions per year for the past 5 years, 1.2 (CV = 0.6) sea lions annually for the GOA groundfish trawl fisheries, and 0.8 (CV = 1.0) for the GOA groundfish longline fishery (Angliss *et al.*, 2001). Resultant ranges of observer coverage relative to fishing effort was 53% - 74% for BSAI groundfish trawl, 33% - 55% for GOA groundfish trawl, and 8% - 21% for GOA groundfish longline fisheries (Angliss *et al.*, 2001). There appears to have been a slight decline in estimated incidental take rates throughout the 1990s. A closer examination shows no apparent 'hot spots' of incidental catch (Figure 3.1-10), nor an apparent relationship between mortality and magnitude of catch. Because of the size class requirements for observer coverage, if vessels with limited or no coverage operate in ways different than the larger vessels, either in technique or area, then these mortality estimates could potentially be biased.

Steller sea lions are incidentally taken by commercial fisheries other than groundfish fisheries, including some nearshore salmon drift or set gillnet fisheries and halibut longline fisheries. An estimated minimum mean annual mortality rate from the past five years of data for all commercial fisheries taking Steller sea lions from the western stock is 28.3 (CV = 0.64) sea lions per year (Angliss *et al.*, 2001). However, many fisheries known to interact with Steller sea lions have not been observed, and thus this should be considered a minimum estimate.

Entanglement of Steller sea lions in derelict fishing gear or other materials seems to occur at frequencies that do not have significant effects upon the population. From a sample of rookeries and haulout sites in the Aleutian Islands, of 15,957 adults observed, Loughlin *et al.* (1986) found only 11 (0.07%) entangled in marine debris, some of which was derelict fishing gear. Observations of sea lions at Marmot Island for several months during the same year observed 2 of 2,200 adults (0.09%) entangled in marine debris. During 1993-97, only one fishery-related stranding was reported from the range of the western stock, a sea lion observed in August 1997 with troll gear in its mouth and down its throat (Angliss *et al.*, 2001). Entanglement of sea lions in derelict fishing gear or other marine debris does not appear to represent a significant threat to the population.

Steller sea lions are primarily utilized for subsistence purposes in communities within the range of the western stock. Pinniped harvests in Southeast Alaska tend to be dominated by harbor seal rather than Steller sea lions, and essentially all of the harvest is from the western stock. Of these, most are harvested in the Pribilof Islands. Estimates of the total number of sea lions taken (harvested plus struck and lost) declined over the six year period of 1992 - 1998 from 549 to 171 per year (Angliss *et al.*, 2001), with an overall mean annual take of 329 sea lions for the entire period.

Harvest levels typically have been lowest during June - August, peaking during September - November and declining through May, but this seasonality has been less pronounced since 1996 with declining harvest rates (Wolfe and Mishler, 1997). The proportion of the harvest comprised of female sea lions has been relatively low. For 1996 - 1998, adult females comprised 14.2%, 9.2%, and 6.9% of the total harvest, while juvenile females accounted for 5.8%, 6.9% and 3.0% (Wolfe and Mishler, 1997; Wolfe and Hutchinson-Scarborough, 1999). Takahashi and Wada (1998) used a modified Leslie matrix model to assess the possible effect of hunting Steller sea lions in Japanese waters and concluded that hunting near Hokkaido to reduce damage to local fisheries likely depleted the sea lion population in the Kuril Islands.

Illegal shooting occurs, but the frequency of occurrence is difficult to estimate. NMFS successfully prosecuted two cases of illegal shooting of sea lions in the Kodiak area in 1998, and two cases in Southeast Alaska between 1995 - 1999 (Angliss *et al.*, 2001).

Based on a published life table and the current rate of decline, Loughlin and York (2001) estimate the total number of mortalities of non-pup Steller sea lions in 2000 was about 6,425 animals; of those, 4,710 (73%) were mortalities that would have occurred if the population were stable, and 1,715 (27%) were additional mortalities that fueled the decline. They tabulated the levels of reported anthropogenic sources of mortality (subsistence, incidental take in fisheries, and research), guessed at another (illegal shooting), then approximated levels of predation (killer whales and sharks). They attempted to partition the various sources of "additional" mortalities as anthropogenic and as additional mortality including some predation. Loughlin and York (2001) classified 438 anthropogenic mortalities and 779 anthropogenic plus some predation mortalities as "mortality above replacement;" this accounted for 25% and 45 % of the estimated total level of "mortality above replacement." The remaining mortality (75% and 55%, respectively) was not attributed to a specific cause and may be the result of nutritional stress.

3.1.1.10 Natural Predators

A brief review of predation on Steller sea lions by killer whales and sharks was presented in the 2000 BSAI/GOA groundfish BIOP (NMFS, 2000). Based on surveys of researchers, fishers, tour boat operators and others, more lethal interactions may occur in the Aleutian Islands compared to other parts of Alaska (Barrett-Lennard *et al.*, 1995). In a study dedicated to tracking killer whales in Prince William Sound during 1984 - 1996, 31 documented marine mammal kills by transient killer whales none were of Steller sea lions (Saulitas *et al.*, 2000). However, nearly 33 (14/43) observed harassments of Steller sea lions by PWS or GOA transient killer whales were observed. Observations were conducted during summer months, and the availability of juvenile sea lions to killer whales may be higher in this area during spring months, when sea lions arrive seeking herring. Based in part on these observations, and on stomach contents of six stranded killer whales, sea lions were estimated to comprise 5% - 20% of killer whale diet (Matkin *et al.*, 2001). Expanding this to account for daily killer whale metabolic needs, average size and caloric content of sea lions consumed, and a population estimate of killer whales, a range of the percent of sea lion mortalities attributable to killer whales was estimated to be 6% - 77%, with a best estimate of 27% (Matkin *et al.*, 2001). The results of this exercise highlighted the need for improved data on killer whale population size and proportion of sea lions in their diet, and suggests that killer whale predation may be a factor in current decline and lack of recovery of sea lions. (Estes *et al.*, 1998; Matkin *et al.*, 2001).

Attacks by great white sharks have been documented on sea lions at the southern end of their range in California (Ainley *et al.*, 1985). Though Alaska waters lie mostly north of the theorized normal range of white sharks, sleeper sharks (*Somniosus pacificus*) range throughout the Gulf of Alaska and Bering Sea, and small marine mammals have been documented in stomach contents (Yang and Page, 1999). One such occurrence was the discovery of harbor seal remains from one shark taken in Alaska (Bright, 1959), though whether this seal was predated or scavenged is unclear. There have also been recent reports of sleeper sharks collected in Alaska containing cetacean and harbor seal remains (Hulbert *et al.*, 2001). No remains of Steller sea lions were found in 13 sleeper shark stomachs collected in the Gulf of Alaska between June and August 1996 in areas near active sea lion rookeries and haulout sites (Yang and Page, 1999). Though this is a small sample size, the stomach contents indicated primarily benthic feeding, despite the proximity and local abundance of sea lion pups (Yang and Page, 1999). Pinniped residues were found in the stomach of 1 sleeper shark among 148 sampled from the Bering Sea in 1997 (Orlov, 1999).

3.1.1.11 Disease and Contaminants

As with any wild mammal population, a multitude of infectious (viral, bacterial, parasitic, or mycotic) or toxicological (heavy metal, organochlorine) diseases may afflict Steller sea lions. Many anatomical and clinical studies have been performed to determine disease prevalence, with an ultimate goal of determining incidence, interactions with environment, and what role disease may play in the population decline or as an impediment to recovery.

Infectious

Many diseases common to otariids in general and sea lions specifically can cause reproductive failure or death, and have thus been considered relative to their role in the population decline (NMFS, 1995). Among those potentially pathogenic that have tested positive for exposure in some sea lions are calicivirus (San Miguel sea lion virus; Barlough *et al.*, 1987), *Listeria* sp. (Spraker and Bradley, 1996), *Edwardsiella tarda* (Spraker and Bradley, 1996), *Bordetella bronchiseptica* (Spraker and Bradley, 1996), canine distemper virus, phocine distemper virus, phocid herpesvirus 1, *Salmonella* sp. (Spraker and Bradley, 1996), *Toxoplasma gondii*, and chlamidia (Sheffield and Zarnke, 1997). Prevalence or isolation of pathogens occurs throughout the range, with no immediate temporal/spatial pattern detectable due largely to small or infrequent sampling (Sheffield and Zarnke, 1997; NMFS, 1995). No exposure to influenza A or *Brucella* spp. was detected (Sheffield and Zarnke, 1997).

Parasitic

Calkins *et al.* (1994) reported finding numerous lesions in adult and juvenile sea lions necropsied during the *Exxon Valdez* oil spill. Gross lesions caused by parasites were found in the nasal cavity, stomach, and intestine, and were unrelated to hydrocarbon exposure (Calkins *et al.*, 1994).

Nasal mites infect sea lions in Alaska (Fay and Furman, 1982) and Russia (Konishi and Shimazaki, 1998) by at least two years of age, though nasal mites and sea lions have apparently evolved into a relatively neutral, or benign, relationship (Konishi and Shimazaki, 1998).

Contaminants

Organic and inorganic chemicals from pesticides and industrial applications that accumulate in food webs and are hazardous to wildlife include persistent organic pollutants (such as DDT, PCBs, chlordane, hexachlorocyclohexane, dioxin), heavy metals (lead, cadmium, mercury), radioactive elements or compounds, and petroleum hydrocarbons. Contaminants can be transported to Alaska via atmospheric or oceanic currents, or can be found in localized point sources such as abandoned military installations, industrial complexes, mining sites, land or sea dumps, and from discharges or spills (AMAP, 1997). Contamination of wildlife can result from inhalation, absorption through skin, direct ingestion, or by consumption of contaminated prey (Marine Mammal Commission, 1999). Changes in diets or ecosystem trophic webs can thus affect the contaminant burden of top predators (Marine Mammal Commission, 1999). Toxic effects of contaminants in wildlife and marine mammals have been associated with reproductive failures (Helle *et al.*, 1976; Reijnders, 1986), population declines (Martineau, 1987), carcinomas (Gulland *et al.*, 1997), and immune suppression (deSwart *et al.*, 1995; Ross *et al.*, 1996).

Few analyses have been published on heavy metal contamination in Steller sea lions. Evidence of transitory metals accumulation in Southeast Alaska sea lions was found by Castellini and Cherian (1999). They found that circulating zinc (Zn) and metallothionein (a chelating compound) were increased in Southeast Alaska sea lion pups during the early 1990s, and returned to values comparable to Aleutian Island pups by 1997. Zinc, copper, and metallothionein levels were comparable between pups sampled from the western stock and Forrester (Lowrie) Island (eastern stock), and lower than captive sea lions (Castellini and Cherian, 1999). Hepatic metal concentrations in Steller sea lions have generally been much lower than found in northern fur seals (Noda *et al.*, 1995). Vanadium concentrations in Steller sea lion livers ranged between 0.023-0.43 µg/g of wet weight, and positively correlated with levels of selenium, silver, and mercury (Saeki *et al.*, 1999).

No toxicological studies have been performed on otariids to determine clinical ramifications of increasing contaminant burdens. However, Organochlorines have been associated with levels of health concern in other animals. Mink kit survival was compromised at approximately 8,000 ng/g lipid weight (AMAP, 1997), immunosuppression in harbor seals was detected at average concentrations of 16,488 ±1023 ng/g lipid weight, and premature parturition in California sea lions was observed at burdens of 134,000 ng/g lipid weight (DeLong *et al.*, 1973).

Lee *et al.* (1996) examined Steller sea lion blubber and liver samples from the Bering Sea and Gulf of Alaska and found that blubber PCBs ranged 5,700-41,000 ng/g lipid in males, and 570-16,000 ng/g lipid in females. PCB concentration in males was orders of magnitude higher than other Arctic and Alaskan pinnipeds. DDTs in levels in males ranged from 2.8 to 17 ng/g lipid and in females from 0.19 to 6.5 ng/g lipid. For males and females aged 6 and 8 years of age, DDE levels were 5.4 and 1.8 ug/g lipid wt, respectively. Females were found to decrease the contaminant burden throughout life, relative to adult males, by dumping contaminants through lactation. Varanasi *et al.* (1992) obtained sea lions samples from the Bering Sea, Barren Islands, Prince William Sound, and St. George Island (Pribilof Islands) and found organochlorine levels in the blubber at 23000 +/-37000 ng/g, wet weight. There was large variance because of the large range of 1,800-110,000 ng/g. The high level at 110,000 ng/g was from a 1-2 year old male from the Bering Sea.⁴ Finally, the NMFS Northwest Center examined blubber samples from 24 Steller sea lions from Southeast Alaska and report PCB levels of 630-9900 ng/g wet weight and DDT levels of 400-8200 ng/g wet weight, respectively (NMFS, unpublished)⁵. These studies indicate burdens are present in Steller sea lions that could be sufficient to produce health effects.

Concerns exist that the toxicity of contaminants may increase within an individual in negative energy balance, or nutritional limitation, as lipophilic contaminants such as PCBs are released as blubber stores are metabolized. While levels of circulating organochlorines did increase in the blood of harbor seals with high body burdens of organochlorines fasting for 15 days, immunological responses remained within normal ranges suggesting short-term fasting did not add an additional threat (De Swart *et al.*, 1995). Based on endocrine responses, however, seals with high levels of contaminants were likely to be less likely to adequately respond to stressful situations (De Swart *et al.*, 1995).

Sea lions exposed to oil spills may become contaminated with polycyclic aromatic hydrocarbons (PAHs) through inhalation, dermal contact and absorption, direct ingestion, or by ingestion of contaminated prey (reviewed in Albers and Loughlin, in press)). After the *Exxon Valdez* oil spill in March 1989, Calkins *et al.*

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⁵D. DeMaster, National Marine Mammal Laboratory, 7600 Sand Point Way NE, Seattle, WA 98115.

(1994) recovered 12 sea lion carcasses from the beaches of Prince William Sound, and collected an additional 16 sea lions from haulout sites in the vicinity of PWS and the Kenai coast. The highest levels of PAHs were in sea lions found dead following the oil spill. Sea lions collected seven months after *Exxon Valdez* oil spill of 1989 in Prince William Sound had levels of PAH metabolites in the bile consistent with exposure and metabolism of PAH compounds (Calkins *et al.*, 1994). However, since lesions associated with hydrocarbon contamination were not found in histological exams of any sea lion, there was no evidence of oil toxicity damage (Calkins *et al.*, 1994).

Disease has not been considered to have played a significant role in the overall decline of the western stock of Steller sea lions (NMFS, 1995), but it is inconclusive to what extent it played as a contributory factor, and to what extent disease may be operating as a limitation to recovery. In declining populations, decreased genetic diversity and synergistic effects from chemical contaminant toxicity can act to compound factors that lead to reduced fitness (Bickham *et al.*, 2000).

Field Studies of Health and Condition

Several field studies have been conducted to assess health and condition of Steller sea lions utilizing hematological and morphometric indices of condition (see Donnelly and Trites, 2000 for a review). In general, health assessments based on blood profiles represent physiological responses to relatively recent effects, and thus are most useful for addressing questions about current health status. A limitation of assessing individual health based on a single sample is that without additional observations or sample gathering, it is difficult to generate a definitive diagnosis of a disease state should the indices indicate an abnormality. However, when multiple individuals are combined it may be possible to generate a picture of health status at a population level, a method commonly applied in domestic and wildlife medicine (Seal *et al.*, 1978; Franzmann, 1985), and has been applied to Steller sea lions (Bishop and Morado, 1995; Castellini *et al.*, 1993; Rea *et al.*, 1998). However, the utility of this type of analysis is highly dependent upon the type and scope of factors operating on the population, and on the sensitivity of the measured variables to those factors.

A key to blood chemistry profiling is that metabolites present in the blood can be directly related to nutritional status as well as many other disease states, and can thus present a snapshot of health for animals sampled from a haulout or rookery. Blood chemistry profiles have in general not detected significant health issues for newborn to 1 month old Steller sea lion pups throughout their range in Alaska during the 1990s (Castellini *et al.*, 1993; Bishop and Morado, 1995; Rea *et al.*, 1998). Rea *et al.* (1998) did find metabolites in pups from Southeast Alaska indicative of longer fasting periods than pups from the Aleutian Islands or Gulf of Alaska. This was consistent with longer periods between suckling associated with longer maternal foraging trips (Brandon *et al.*, 1996). However, levels of plasma haptoglobin, an acute phase reaction protein that increases in response to infection, inflammation, trauma and environmental stress, were found to be significantly elevated in pups and adults during 1992-1994 from the Aleutian Islands and Gulf of Alaska compared to those in Southeast Alaska (Zenteno-Savin *et al.* 1997). This could indicate that some stressors were operating on populations in the areas of decline differently than on the area of stability.

Comparisons of growth measurements, such as mass or length at age, are more reflective of longer term conditions experienced by an animal. Steller sea lions sampled in the 1980s weighed less and were shorter for age than sea lions sampled during the 1970s (Calkins *et al.*, 1998), and were less massive than expected based on length-girth relationships (Castellini and Calkins, 1993). These differences were most notable among animals less than 10 years old (Calkins and Goodwin, 1988), and may have been declining since the

1960s (Calkins *et al.*, 1998). These changes are consistent with nutritional limitation. Recent comparisons of body size across regions of decline and stability do not recapitulate the long-term trend, however. There is evidence for larger pup sizes in areas of decline (Rea, 1995; Merrick *et al.*, 1995; Adams, 2000; Fadely and Loughlin, 2001), arising from differential growth rates (Brandon and Davis, 1999). Adult females with pups were not different in size between the regions of stability and decline (Davis *et al.*, 1996), though this sample of unknown age females may not be representative of the populations as a whole.

3.1.1.12 Natural Competitors

Steller sea lions forage on a variety of marine prey that are also consumed by other marine mammals (e.g., northern fur seals, harbor seals, humpback whales), marine birds (e.g., murre and kittiwakes), and marine fishes (e.g., pollock, arrowtooth flounder). To some extent, these potential competitors may partition the prey resource so that little direct competition occurs. For example, harbor seals and northern fur seals may consume smaller pollock than Steller sea lions (Fritz *et al.* 1995). Competition may still occur if the consumption of smaller pollock limits the eventual biomass of larger pollock for sea lions, but the connection would be difficult to demonstrate. Such competition may occur only seasonally if, for example, fur seals migrate out of the area of competition in the winter and spring months. Similarly, competition may occur only locally if prey availability or prey selection varies geographically for either potential competitor. Finally, competition between sea lions and other predators may be restricted to certain age classes, because diet may change with age or size.

3.1.1.13 Influence of environmental and climatic change on Steller sea lions

From 1940-1941 an intense Aleutian Low was observed over the BSAI, and GOA, this was followed from December 1976 to May 1977 with an even more intense Aleutian Low. During this latter period, most of the North Pacific Ocean was dominated by this low pressure system which signaled a change in the climatic regime of the BSAI, and GOA (NRC, 1996). The system shifted from a “cold” regime to a “warm” regime that persisted for several years. Since 1983, the GOA and Bering Sea have undergone different temperature changes. Sea surface temperatures in the GOA were generally above normal and those in the Bering Sea were below normal. The temperature differences between the two bodies of water have jumped from about 1.1° C to about 1.9° C. Recent evidence now indicates that another regime shift occurred in the North Pacific in 1989 (NRC, 1996).

Most scientists agree that the 1976/77 regime shift dramatically changed environmental conditions in the BSAI and GOA (Benson and Trites, 2000). However, there is considerable disagreement on how and to what degree these environmental factors may have affected both fish and marine mammal populations. Productivity of the Bering Sea was high from 1947 to 1976, reached a peak in 1966, and declined from 1966 to 1997. Some authors suggest that the regime shift changed the composition of the fish community and reduced the overall biomass of fish by about 50 percent (Merrick *et al.*, 1995; Piatt and Anderson, 1996). Other authors suggest that the regime shift favored some species over others, in part because of a few years of very large recruitment and overall increased biomass (Beamish, 1993; Hollowed and Wooster, 1995; Wyllie-Echeverria and Wooster, 1998).

It is reasonable to conclude that the regime shift created environmental conditions that produced very large year classes of gadids (i.e. pollock and Pacific cod). However, because of the historically high catches of gadids before the regime shift occurred, it is not likely that the regime shift favored gadids in a way which

would allow them to out compete other fish species and dominate the ecosystem, although the absolute level of biomass is not well known.

Many competing factors have contributed to the ecosystem in which Steller sea lions now depend Pauly *et al.*, 1998). However, the important question is whether the diet of Steller sea lions was adversely affected by the regime shift. Specifically, the question has been raised as to whether the increase in pollock abundance is now contributing to the decline of Steller sea lions. From the information available, it seems reasonable to conclude that gadids (i.e., pollock and Pacific cod) were abundant before the regime shift, and that sea lions relied upon them for food before the decline. Therefore, it is unlikely that a change in the structure of the ecosystem, resulting in a dominance of gadids is the sole cause of the current decline.

Shima *et. al.* (2000), looked at the GOA and three other ecosystems which contained pinniped populations, similar commercial harvest histories, environmental oscillations, and commercial fishing activity. Of the four ecosystems only the GOA pinniped population (Steller sea lions) were decreasing in abundance. They hypothesized that the larger size and restricted foraging habitat of Steller sea lions, especially for juveniles that forage mostly in the upper water column close to land, may make them more vulnerable than other pinnipeds to changes in prey availability. They further reasoned that because of the behavior of juveniles and nursing females, the entire biomass of fish in the GOA might not be available to them. This would make them much more susceptible to spatial and temporal changes in prey, especially during the critical winter time period (Shima *et. al.*, 2000).

Figure 3.1-1 Steller Sea Lion Western Stock Population Trends, 1976-2000.

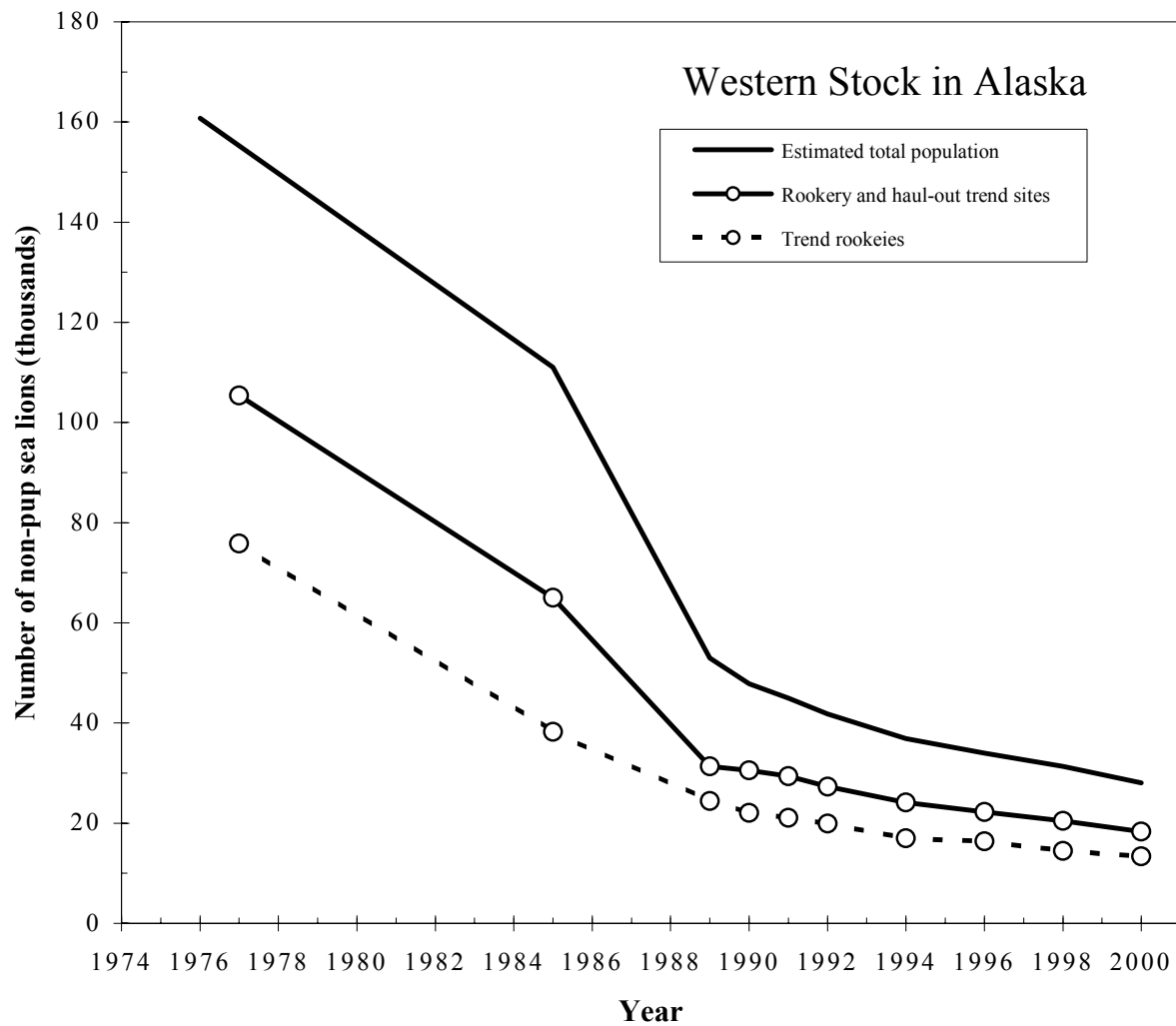


Figure 3.1-2 Regional Steller Sea Lion Population Trends, 1976-2000.

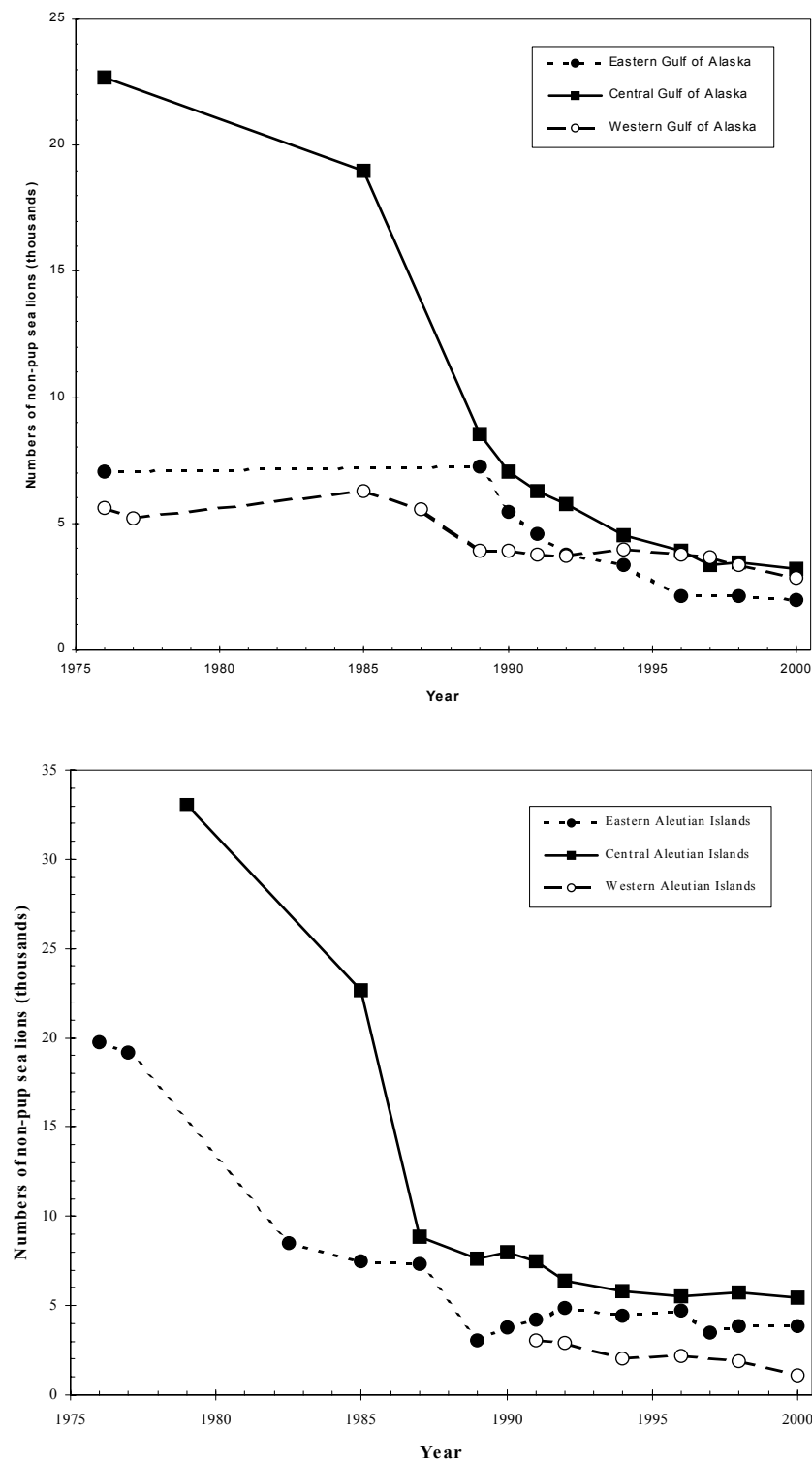


Figure 3.1-3 Counts of Steller Sea Lion Pups in Alaska (Sease and Loughlin, 1999).

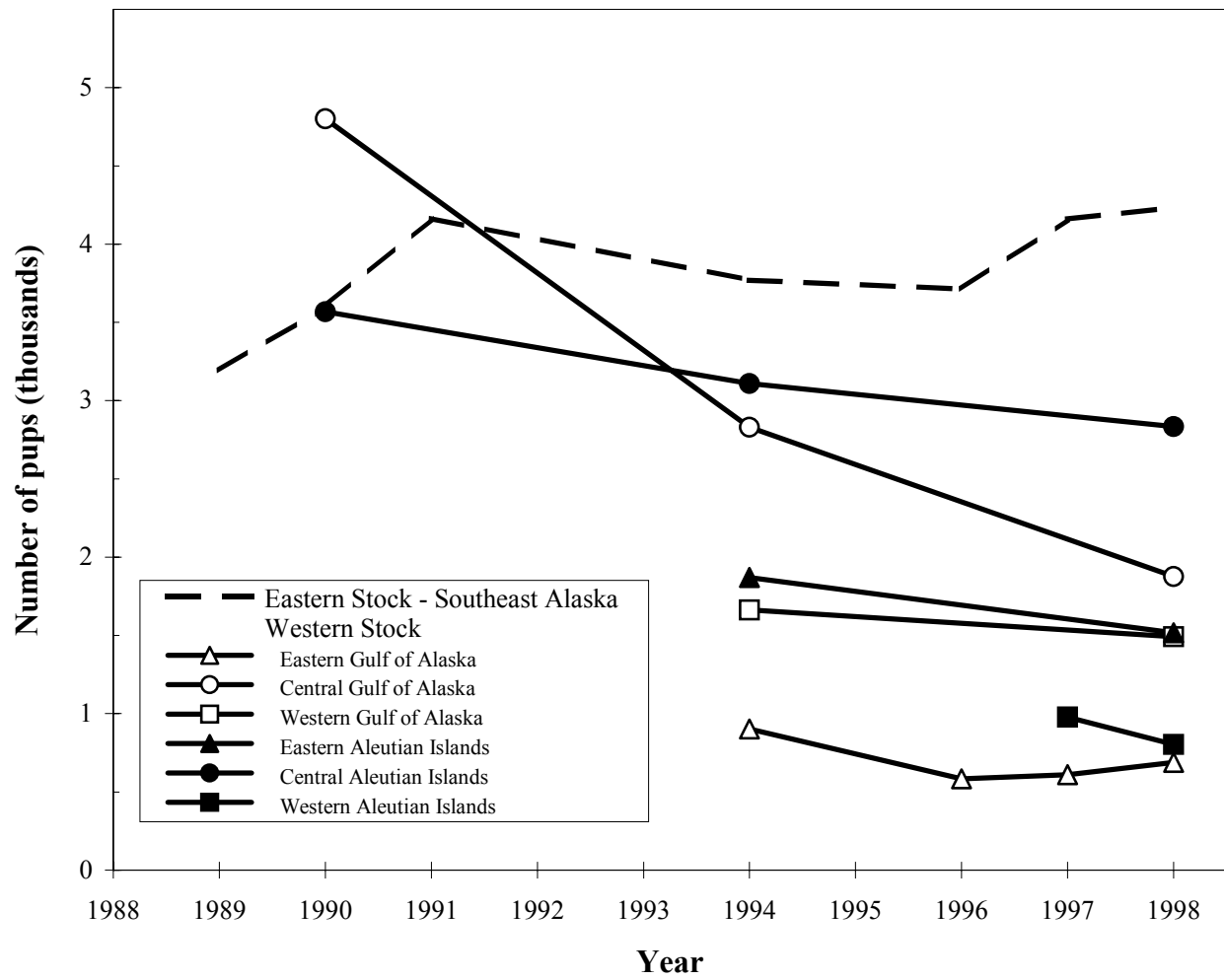


Figure 3.1-4 Population Trends of Steller Sea Lion Eastern Stock in Southeast Alaska, 1975-2000.

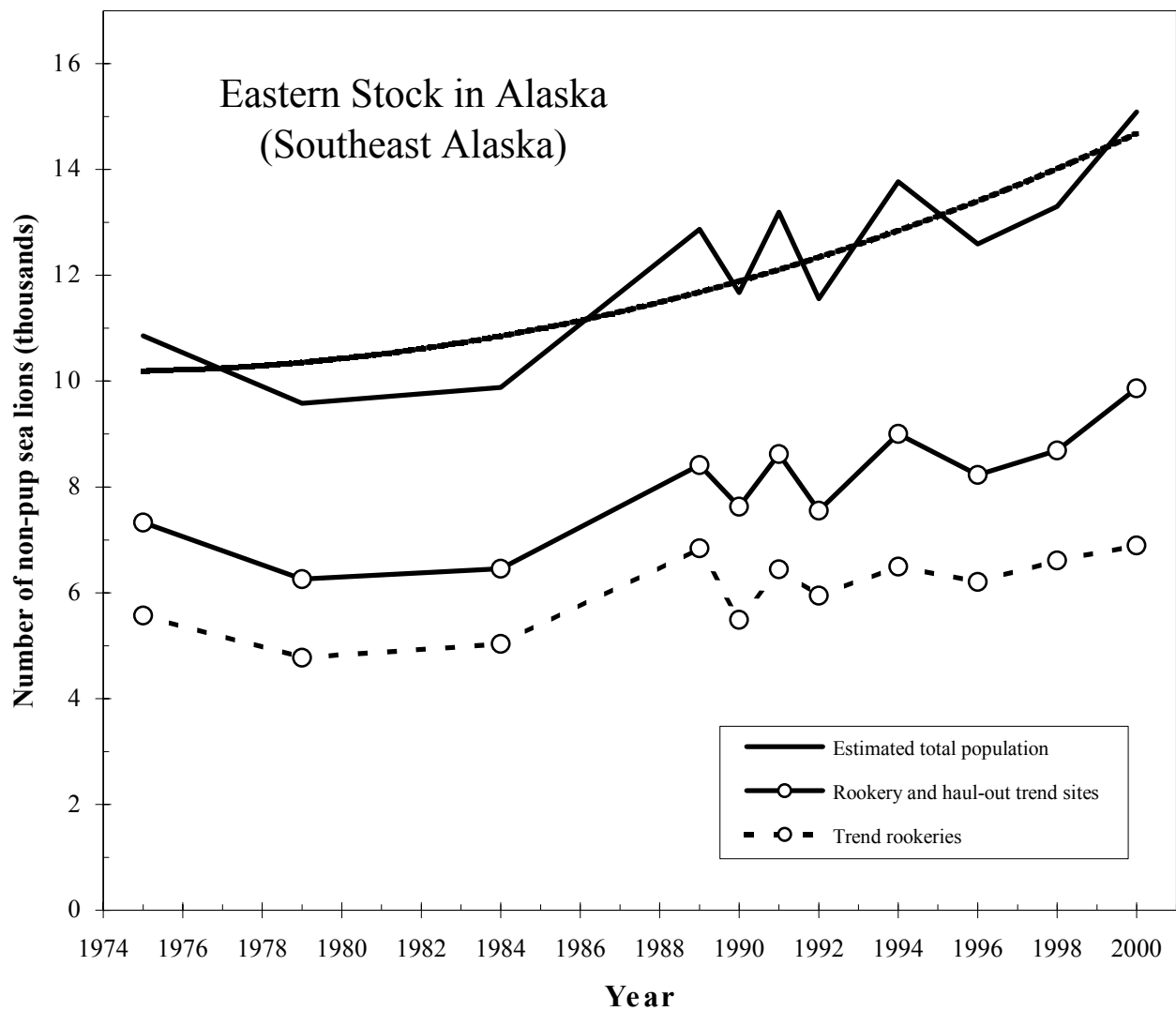


Figure 3.1-5 Counts of Steller Sea Lions in the Eastern Stock, 1982-1998 (adapted from Angliss *et al.*, 2001).

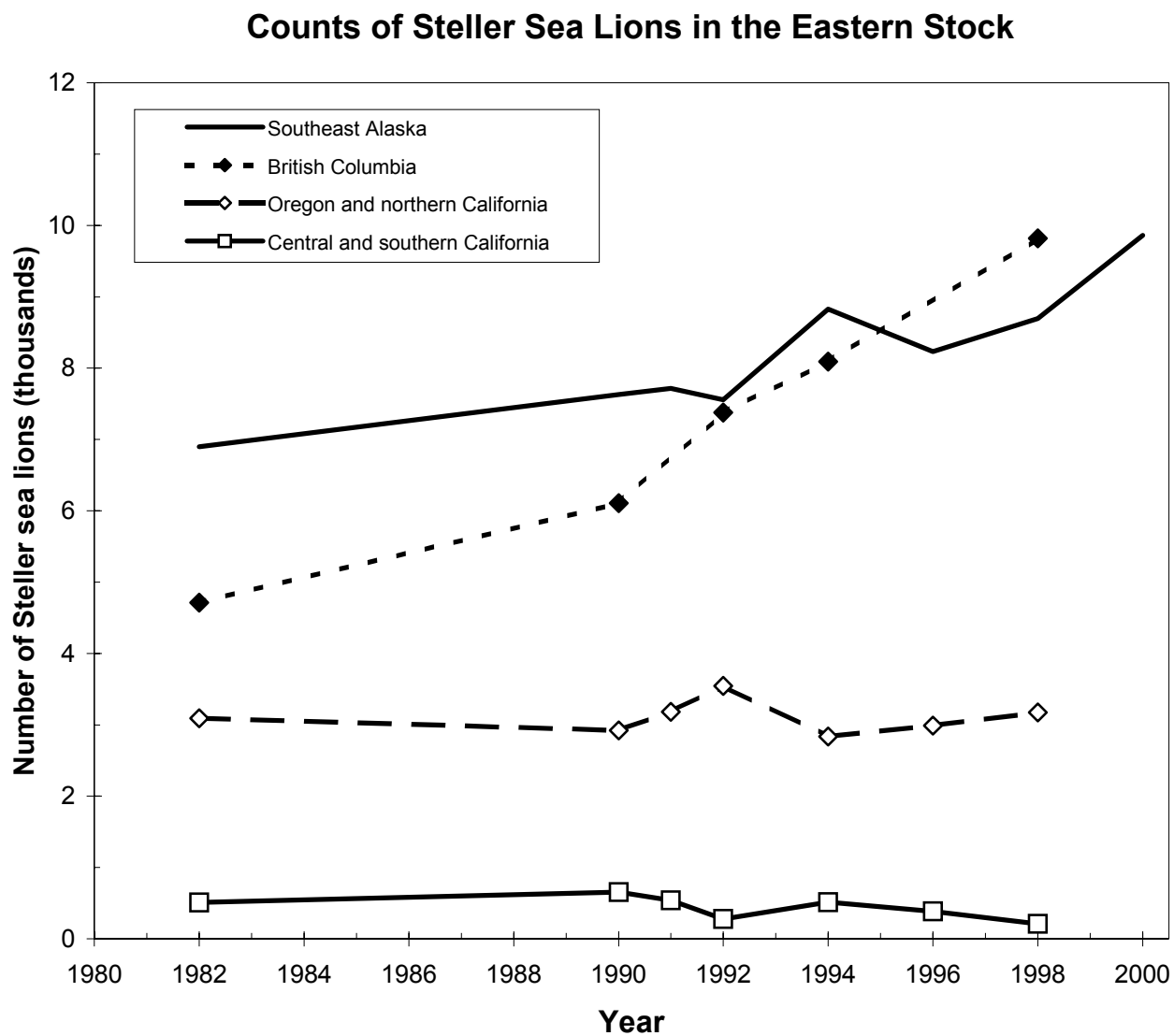


Figure 3.1-6 Distribution of Steller Sea Lions in the Bering Sea and Western/Central Gulf of Alaska From Platform of Opportunity (POP) Database (NMFS Data).

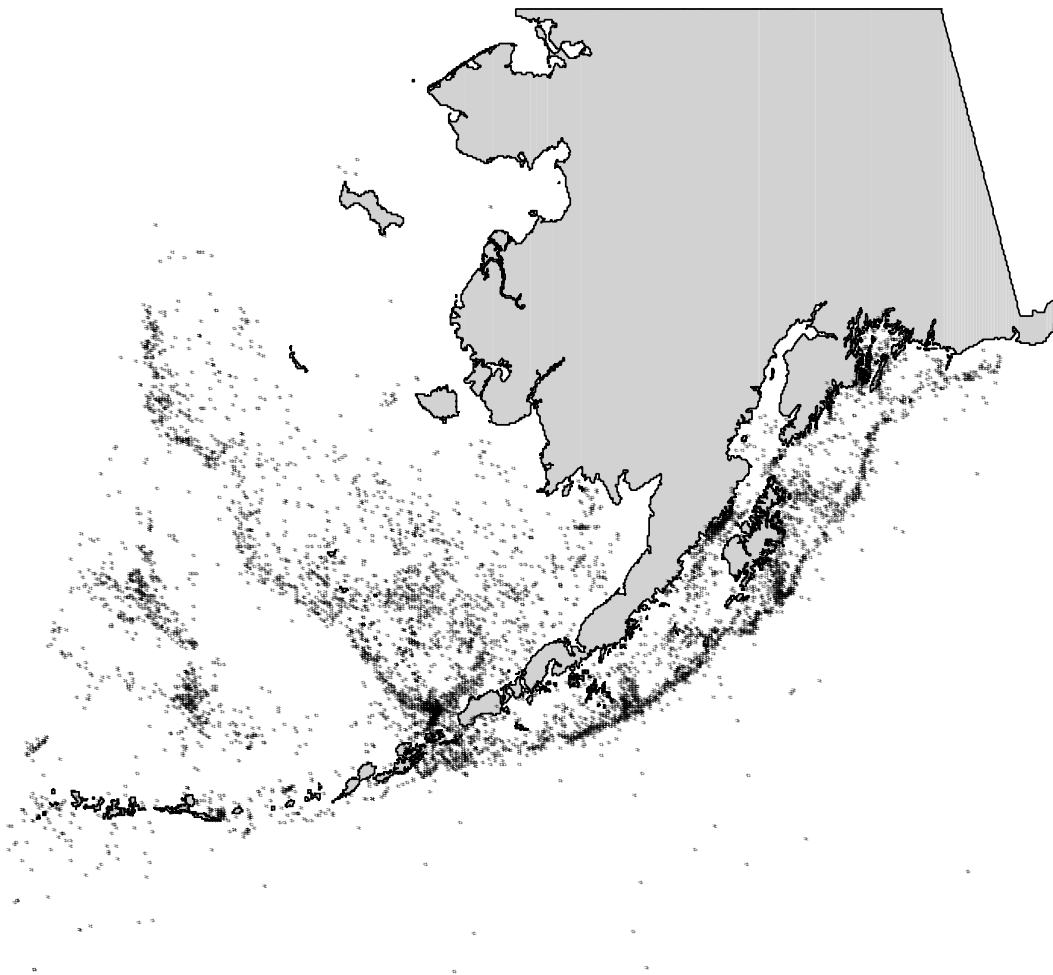


Figure 3.1-7 Proportion of Dives by Depth Range for Young-of-the-year (WYOY) and Adult Female Steller Sea Lions in Summer (SAF) and Winter (WAF) Tracked During 1990-1993 (Merrick and Loughlin, 1997).

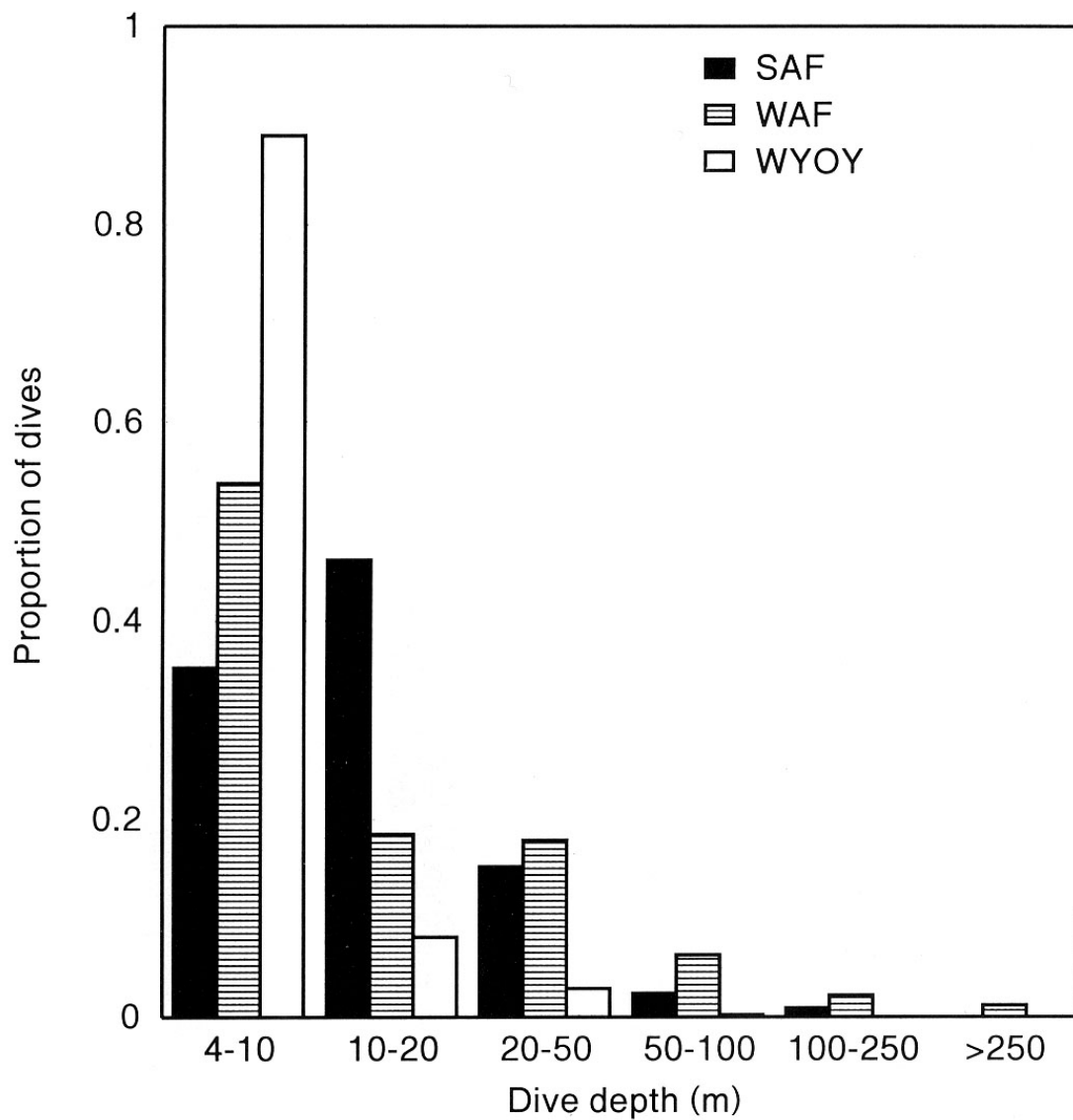


Figure 3.1-8 Frequency of Occurrence of Prey Items Occurring in Steller Sea Lion Scats, in All Regions and Seasons, 1990-1998 (Sinclair and Zeppelin, submitted).

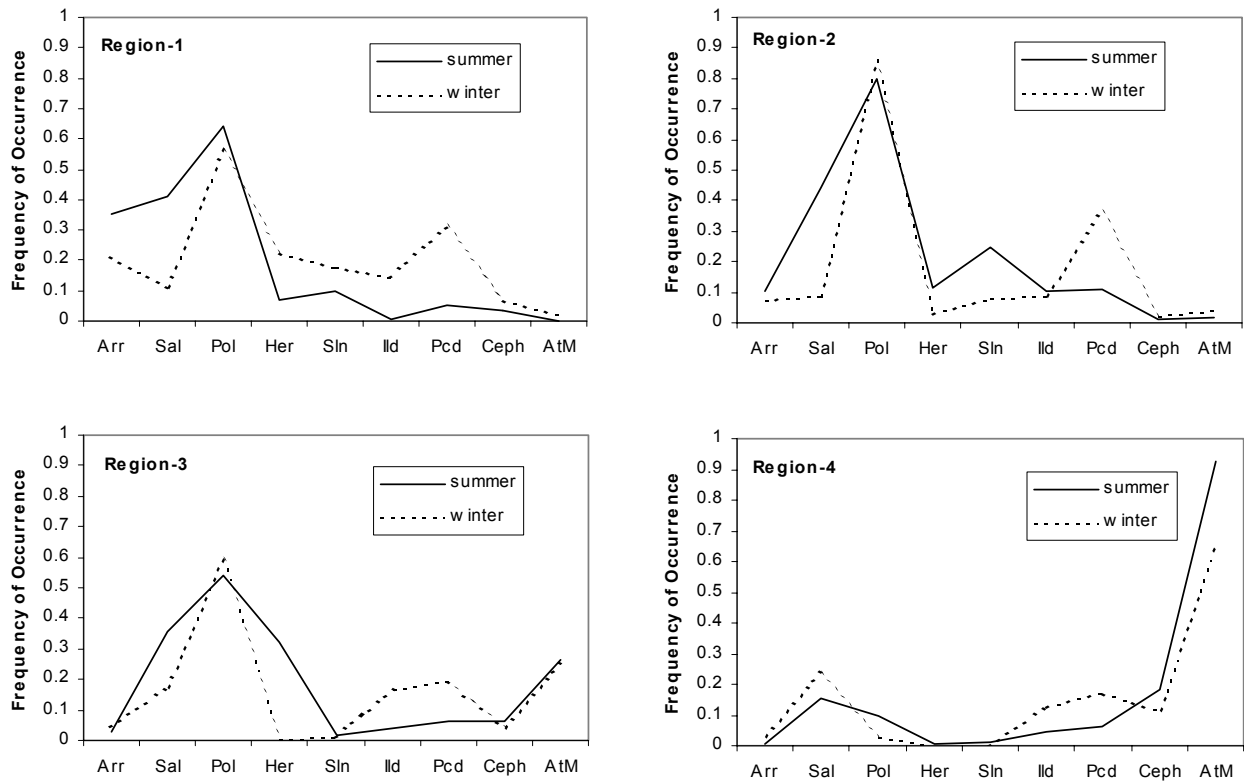


Figure 3.1-9 Steller Sea Lion Diet Divisions in Relation to Population Trends (1989-1994) (Sinclair and Zeppelin, submitted).

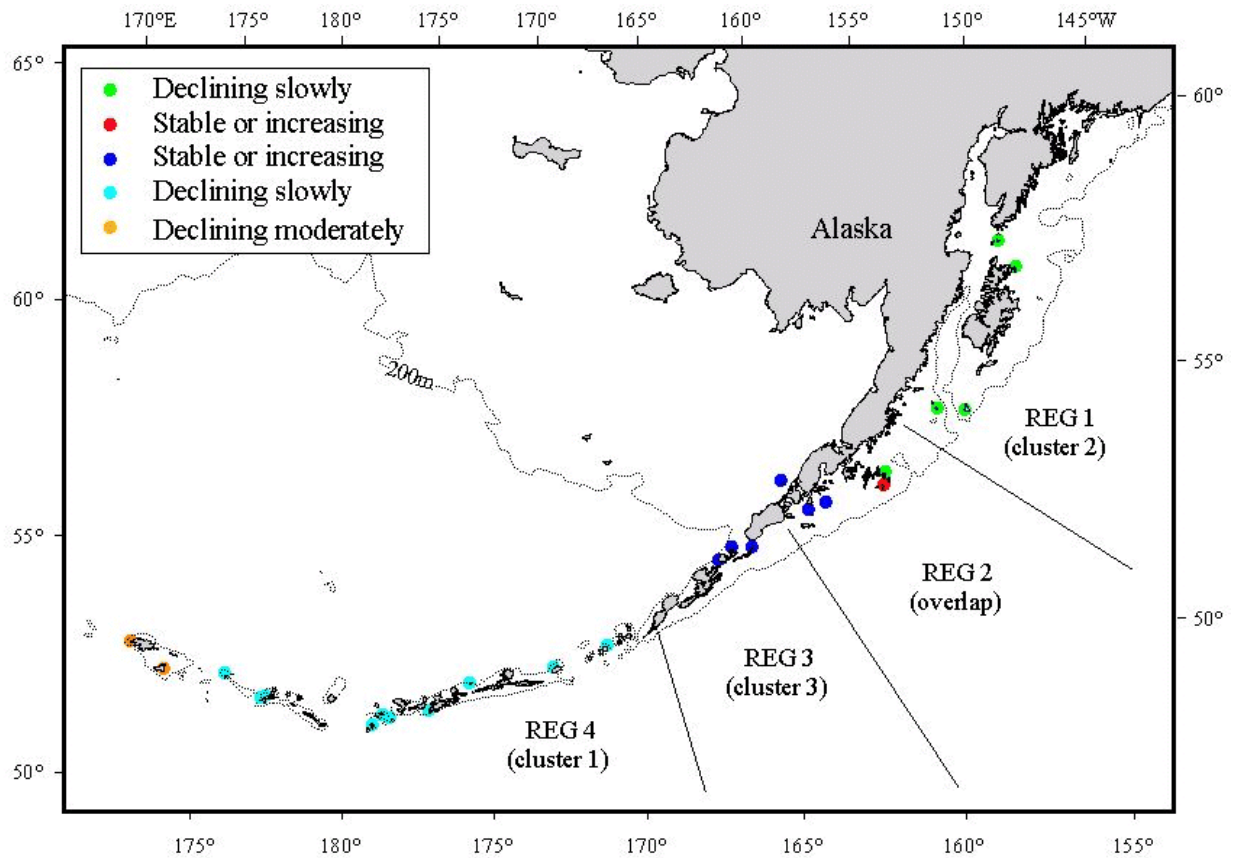


Figure 3.1-10 Locations of Observed Steller Sea Lions Incidentally Caught in Domestic Groundfish Trawl (circles) and Longline (triangles) Fisheries, 1989-1999.

